THE APPLICATION OF ECONOMICS TO WATER MANAGEMENT IN SOUTH AFRICA

R.I. MIRRILEES S.F. FORSTER
AND C. J. WILLIAMS

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by

R.I. MIRRILEES

S.F. FORSTER

C.J. WILLIAMS

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1. INTRODUCTION

1.1 PROJECT BACKGROUND

The research project that led to the production of this document had the original aim of evaluating the applicability of economic principles[1] to decision-making in water management in South Africa.

The reasons for wanting to do this were twofold.

Firstly, in 1970, the Commission of Enquiry into Water Matters strongly advocated that

> *"overall planning of the use of South Africa's limited water resources should be thoroughly integrated with the economic planning of the country as a whole, taking socio-economic conditions into account ..."*

Furthermore the Department of Water Affairs and Forestry stated in its 1986 publication "The Management of the Water Resources of the Republic of South Africa" that:

> *"The national management strategy aims at optimising national benefits from all uses of water according to a scarce resource allocation strategy, implemented in harmony with the prudent development of the water infrastructure and the application of appropriate controls".*

Prior to this present study, however, little attention had been paid to water management issues from the broad economic perspective.

Secondly, it was known that a considerable body of work linking water management and economics had been performed internationally during the eighties, and it seemed logical to consider whether there were lessons to be learnt from this by local water managers.

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1. In the original formulation of the project, reference was made to "resource economics" rather than "economic principles". It was soon realised, however, that this would be too restrictive a topic and that we first needed to consider economics in general before focussing on specialist fields..

Two broad "philosophies" guided the research. The first was that the quest was for useful and generally-applicable methodologies - not for solutions to specific practical problems being experienced by South Africa's water managers at present. The second philosophy qualified this by requiring that the project should be problem-driven. For this purpose, two geographical areas of study were originally selected to serve as starting points, demonstrating respectively water "quantity" and "quality" problems. However, many other problems were also identified in the course of the study through discussions with people involved in or affected by water management. These were at least as influential in shaping the course of the study as were the aforementioned areas of study themselves[2].

To achieve the aim of the project, it was proposed to develop an overview of the application of economic principles to water management in other parts of the world; and identify, analyse and evaluate potential areas for the application of these principles to decision-making in water management in South Africa.

1.2 THE BRIEF

The brief provided the researchers with the opportunity to develop statements of broad economic principle, which after further investigation and debate, could lead to their adoption in finding solutions to water management problems. If we believed that economics had something to contribute to water management, here was the chance to say so clearly and forcefully, without becoming embroiled in detailed technical issues. This allowed us to step back from the minutiae of economic theory in order to consider where we believed economics' most significant potential contributions to lie. After some false starts, we concluded that these are in providing:

- an alternative (to the traditional supply-fix) macro management approach to deal with water quantity (allocation) problems;

2 In fact, neither case-study is discussed explicitly in this document. The two case-studies originally selected were the Mgeni and Sabie-Sand catchments. However, it was soon realised that becoming immersed in the problems of those catchments detracted from the primary aim of focusing on the application of economics to water management in South Africa in general. Although both these case studies did contain many of the issues, such as water allocation between competing users, and water pollution from communities that did not possess the resources to remedy the situation.

- an alternative (to the traditional command-and-control) macro management approach to deal with water quality (pollution) problems; and
- methods to assist in the piecemeal implementation of macro-economic approaches, whether these be the ones we advocate or the more traditional ones.

These potential contributions are reflected in the format of the Appendices. The difficulties were the obverse of the opportunities: strong statements were not always supported in the literature, although it was realised from an early stage that the methods for applying economics to water management in South Africa would have to be tailored specifically to the problems encountered in this part of the World. Also, it was realised that making strong statements tended to encourage a selective approach to the topic.

As for the first of these difficulties, the literature offers considerable support for our positions in the case of the macro water quality approach and the methods to aid piecemeal implementation of macro approaches. Both of these topics have been discussed extensively in the literature and there is wide consensus in the economics profession about how they should be tackled. Consequently, the discussion that we provide in this document is fairly concise.

Unfortunately, and somewhat to our surprise, we found that we could not place equal reliance on the literature in the case of our recommended approach to macro water quantity management. This approach is, in essence, the obvious one of using prices to ration this "scarce" resource. The quintessential advantage of proper pricing is that it will prevent water scarcities from arising. The most basic of economic theory shows that scarcity is nothing more or less than an imbalance between demand and supply: more specifically a situation in which demand outstrips supply. As a general rule demand is a negative function of price, and supply a positive one. Hence it follows that a rising price will reduce demand and increase supply, so that any "scarcity" can be overcome simply by a price increase of the appropriate magnitude.

Therefore it is understandable that an insistence on the determination of a "correct" price level for water forms a constant refrain in economic analyses of the "water quantity problem". Many discussions can be found in which the central message is that if one gets the price of water "right", the allocation problem will be solved. Obviously there is more to it than that. Water has unusual, perhaps unique, features which place it outside the more usual scope of economic rules and techniques, rendering their glib transferral to this special case inadequate.

Somewhat puzzlingly we were unable to find studies which explored this transferral fully, and clearly we could not leave the argument at that simplistic level. Accordingly an explicit attempt has been made here to rework standard economic theory so as to encapsulate water's peculiar properties. In consequence, the "pricing" approach to allocation receives slightly unorthodox treatment and emerges in a rather unfamiliar guise. Essentially, this boils down to the establishment of quasi-pricing rules to be applied within a predominantly public sector supply framework (what we might call an "as-if-pricing" approach), and necessitates the development of sophisticated market simulation capabilities in order to be practically meaningful.

We believe that this extension of a more customary doctrine makes a real contribution to South Africa's water management.

Turning to our second difficulty, namely that making strong statements encourages a selective approach to the topics dealt with, we must draw the reader's attention to a number of subjects that have attracted extensive debate in the economics literature but have not been discussed in this document. They are:

- The recreational use of water;
- Irrigation schemes as a vehicle for economic development;
- The use of groundwater; and
- Interregional water transfers.

The first of these topics has attracted debate because demands for water for recreational use are beginning to outstrip its supply in some developed countries. As far as is known, this has not as yet become a serious issue facing water managers in South Africa, and for this reason no discussion is provided in this document. An introduction to the subject can be found in Davidson P, Adams FG & Seneca J, "The social value of water recreational facilities resulting from an improvement in water quality" in Kneese & Smith (1965).

The remaining three topics have been excluded from the discussion for a different reason. Although they are very pertinent to water management in South Africa today, they are also complex and wide-ranging. To deal with them adequately would have required additional analyses that went beyond the scope of this present exercise and would only have served to distract our attention from it. We believe that the document that we present here establishes a sound methodological framework that will make these other topics easier to analyse, and recommend them as candidates for future research.

Irrigation schemes have traditionally been viewed as suitable vehicles for attempts to promote economic development. They can also have high opportunity costs as a result of the high water losses on irrigation schemes and the fact that crops can have a low value relative to other uses to which the water could be put. In view of the possibility that the extension of irrigated agriculture could be viewed as politically attractive in the future South Africa, there is an urgent need for an examination of the long-term sustainability of this activity. Introductions to some of the relevant issues can be found in Frederick (1975), Field, Barron & Long (1974) and World Bank literature.

Given that irrigated agriculture is usually both the largest and least-efficient of a country's water-using sectors, and that agriculture is also a source of water pollution, it is inevitable that recommendations aimed at improving water management will have much of their impact on this branch of the economy. The approaches that are recommended in this document are no exception to this rule. Therefore it is unfortunate that we found it to be beyond our scope to include here an analysis of the use of groundwater in agriculture. The two aspects that are of particular interest are the conjunctive use of surface and groundwater resources, and groundwater quality: they are introduced respectively in O'Mara (1988) and Saliba (1985).

Finally, South Africa is already dependent on large-scale interregional water transfers, and seems set to become more so. With the likelihood that regions will become more autonomous in future, the economic implications of these transfers deserve study. An introduction to the relevant issues can be found in Howe (1985).

1.3 REPORT STRUCTURE

The report, as it is presented here, comprises three parts. Part 1 'Water Management Issues" is devoted exclusively to describing a number of broad problems which are frequently encountered in water management undertakings in South Africa. These issues are only partially resolved at present to varying degrees of satisfaction of the decision-makers. In addition to their description, Part 1 also matches the problems with appropriate economic instruments, the details of which are contained in A, B, C and D, to be found in Part 2 of the report.

Appendix A, focuses on water quantity issues such as water pricing and allocation; Appendix B describes the use of economic instruments for managing water quality; and Appendix C comprises a collection of economic instruments used for general evaluation purposes, but which in this instance are presented from the perspective of water management. The instruments described in Appendix C are grouped into two categories; cost-benefit analysis techniques, and those used in valuing the natural environment. Appendix D comprises a detailed bibliography serving both the references contained in the report and appendices, and the further information needs of the reader.

CONTENTS

PART ONE

PREFACE

Despite several policy statements from government during the last 20 years which indicated the importance of considering the economic aspects of water resource development, there has been little attention, prior to this study, paid to analysing water management issues from a broad economic perspective. It was known that a considerable body of work linking water management and economics had been performed internationally during the 1980's and it seemed logical to consider whether there were lessons to be learned from this by South Africa.

Two basic principles guided the research. The first was that the quest was for useful and generally applicable methodologies, and not for solutions to specific problems being experienced by South Africa's water managers. The second principle, however qualified this by requiring that the project should be problem driven. In the course of the project, a great many problems were identified. Several of these possessed a dynamic nature due to the rapidly changing socio-political environment in South Africa. All these problems were instrumental in shaping the course of this study and the final product.

After extensive research, it was concluded that the most promising contribution of economics to South African water management lay in the following areas:

- an alternative (to the traditional supply-fix) macro management approach to deal with water quantity (allocation) problems;
- an alternative (to the traditional command-and-control) macro management approach to deal with water quality (pollution) problems; and
- methods to assist in the piecemeal implementation of macro economic approaches, such as cost-benefit analysis and resource valuation.

As such the content and structure of this report and its appendices reflects these potential applications. Furthermore, the report has a deliberate bias towards the economic aspects of meeting the vast backlog in the water supply and sanitation services to South Africa's many disadvantaged communities.

This document is not aimed specifically at economists. The methodologies it contains will likely prove somewhat conventional to practitioners of this discipline. It is primarily intended for water managers and decision-makers, particularly those who have had limited exposure to economic concepts. Moreover, this report is not a comprehensive manual on the economics of water resource development or its use in water project planning. Although these may be worthwhile products for future consideration, the purpose of this project is to introduce, in broad terms, the potential application of economics to water management in South Africa.

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Part 1

WATER MANAGEMENT ISSUES

2. WATER ALLOCATION [3]

2.1 WHAT ARE THE CONSEQUENCES OF ALLOWING FARMERS TO SELL THEIR WATER?

2.1.1 USA experience

A feature of the USA water rights problem in the last 12 years has been the encouragement of water selling and water rights leasing by farmers to municipal authorities. In several parts of the USA farmers have more water than they need, while the demand from towns and cities keeps growing. To meet this growing demand water supply agencies must look further afield, and the farmers' surpluses suggest themselves as an obvious new source of supply.

This situation was prompted by a number of factors. Firstly, many farmers possessed water rights which exceeded their ability to use the water. Secondly, because of problems of over-production and reduced agricultural subsidies, the profit margins of many farming operations were diminished. Thirdly, attempts by municipal and metropolitan bodies to buy water rights from farmers were met with strong opposition from agricultural bodies, and (where the purchased water was to be exported to a neighbouring state) State officials. Such deals fell victim to protracted litigation.

However, the Reagan Administration recognised the merits of achieving badly needed water redistribution through the market, whilst at the same time improving the economic plight of the farmer. Thus sales of water by farmers to urban water supply agencies were encouraged. However, as municipal users required a high level of assurance of supply, the prospect of farmers changing their minds about short term water sales or even selling their farms to parties not interested in water sales, threatened the success of the initiative. The subsequent leasing of water rights by municipal users for a fixed period with a set date for lease renewal negotiations, provided the necessary security of supply required by the lessee.

3 This section of the report should be read in conjunction with Appendix A.

2.1.2 South African situation

In view of the growing water demand by urban users in South Africa, the question might be asked as to whether the USA experience has application here? Water rights sales are not new to South Africa. Municipalities such as De Aar have for many years bought farms in close proximity to the town in order to abstract the ground water to meet growing urban demands. This practice is sometimes referred to as water farming. Recently South African farmers in the north-western Transvaal considered selling some of their ground water to water scarce municipalities; however, this met with disapproval from organised agriculture which saw it as eroding the agricultural potential of the area.

There are two problems with this type of water sale in South Africa. Firstly, most water rights are granted on the basis of the water being used only for irrigation, stock watering and farm domestic purposes. To re-assign that water to urban/industrial usage would currently require Ministerial permission and consideration by the Water Court. Secondly, the present practice of buying farms in order to obtain the ground water (which is private water and can be used at the discretion of the landowner) can lead to the inefficient use of productive farmland, except perhaps for game farming. It would be more beneficial if several farmers were able to sell their surplus water to a town whilst continuing viable farming operations. This way the municipality would not have to find the capital with which to buy farms, while the farmer benefits from an increased and more secure income from diversified business operations. From a water resources point of view there is the added benefit of water sales providing an incentive to improve on-farm water use efficiency.

Subject to considerations of "rights" to water, as discussed in the next paragraph, arguments derived from economic theory would in general favour the marketability of water, and the institutional changes necessary to make it possible. This is discussed in section 3.2 of Appendix A. There is merit in undertaking a more detailed study in this regard, which would, inter alia, have to give attention to whether or not agriculture deserves superior rights to water than other uses. In the meantime, economic investigation could still contribute to the improvement of water allocation by quantifying the costs and benefits of water farming in order to identify situations in which the reassignment of water to different uses would be beneficial.

2.1.3 Ownership of marketable water

Whilst the arguments for allowing farmers to sell their water may appear convincing, there are certain business and economic objections that have to be overcome in this regard. For example, does the farmer own the water or just the right to use it for agricultural purposes? In other words, is the farmer making money from marketing water (a national asset), the use of which should perhaps be re-allocated by the authorities to meet changing demand patterns. This argument might be extended by claims that many farmers acquired their riparian water rights at a time when the competition for scarce resources was negligible, as happened in the Western part of the USA. Indeed, in some instances this competition might have been reduced artificially through the policy of forced removal of people. Alternatively, the claim might be made that the authorities have failed to timeously review water rights as demand patterns changed, and that now they would be avoiding their responsibility by leaving water reallocation to the market.

Investigating the effects of allocating water via the market mechanism may provide a useful method if maximising water's utility for the public good. At the very least it would be worthwhile to expose the economic option to public debate and challenge the preconception that water must always be allocated by regulation because market forces are inadequate for this task.

2.2 NON-MARKET ALLOCATION : HOW SHOULD SCARCE WATER BE ALLOCATED AMONG COMPETING USERS?

2.2.1 Water allocation problems

A number of South Africa's catchments are approaching, or have already reached, the full economically viable development of the available water resources. The potential for the further impoundment of water in many of these catchments is restricted by the availability of dam sites, reduced catchment yield due to upstream water using activities, and the scarcity of capital needed for large water projects. The question which is posed is, in such situations how are future increases in demand going to be accommodated? Indeed, when one considers the problems currently being experienced in the drought affected areas of South Africa, the question might be rephrased to ask how the existing backlog in demand should be accommodated?

2.2.2 Current water allocation policy

The response given by the Department of Water Affairs and Forestry to the water demand dilemma is that water should be allocated according to the principle of 'Best Joint Utilisation'. This principle is aimed at 'achieving the maximum benefit with minimum detriment to each user group under changing conditions'. In other words it implies some sort of quasi-optimisation aimed at satisfying as much of the justifiable demand as possible within the limitations of the resource base at any point in time.

Although this intention is theoretically sound and meritorious, it is virtually impossible to operationalise. Firstly, in a competitive environment, what is 'best' for one user may incur a loss for another user. Secondly, the term 'joint' can mean anything from a 99% to a 1% apportionment. Thirdly, 'utilisation' can be interpreted as being anything from the satisfaction of basic human needs to the filling of swimming pools. As there is no detailed implementation strategy accompanying this principle, the implication is that users are prepared to compromise with regard to either the use of water currently available to them, or their demands for more water. In this regard, however, water conflicts between sugar farmers and foresters in Natal, and between foresters, irrigators and nature conservationists in the Eastern Transvaal show that this may not be true: the 'haves' jealously guard their current water rights.

To attempt to re-allocate just some of the water which forms part of existing water rights will require a great deal more than good will. Negotiators must possess strong incentives, preferably of an economic nature, before land owners will even consider relinquishing such rights. To an economist, it seems obvious that one of the strongest incentives of all, the maximisation of social welfare, should play a role. Allowing water rights to be traded via the markets would allow this.

2.2.3 Alternative water allocation approaches

Clearly, a specific and realistic objective is required for each and every catchment requiring scarce resource allocation (i.e., where demand outstrips supply) together with a practically achievable implementation methodology. For example, the following objectives can be found in the local and overseas literature, and might be quite realistic objectives for several South African catchments where the available water resources are under severe demand pressures:

(i) A minimum human daily consumptive per capita water allowance of 30 litres.

(ii) A 70% reduction in the occurrence of diarrhoea and related diseases within five years.
(iii) The development of sufficient subsistence irrigation to eradicate nutritional problems and related diseases within five years.
(iv) No further degradation of the natural resource base.
(v) A minimum river flow regime adequate for the maintenance of the aquatic ecosystem, from which only abstractions for basic human needs will be permitted.
(vi) No decrease in the annual value (in real terms) of the agricultural production.
(vii) Maximisation of job opportunities in subsistence and commercial agriculture, i.e. preferential water allocations to the irrigation of labour-intensive crops.
(viii) Restrict water use by afforestation to 10% of the mean annual runoff and 1% of the normal (low) flow.
(vii) Minimise the dependence of the catchment's water users on State financial assistance, i.e. promote economic self-sufficiency.

The above is not just a list of water management objectives. It is in fact a list of catchment development objectives, which must be determined prior to allocating scarce water resources. A problem with such lists, of course, is that there are trade-offs between the various objectives which have to be made to attain an optimum allocation pattern. Often such trade-offs are made by decision-makers on the basis of political preference or accepted regional economic development plans. However, the decision-maker seldom has any way of knowing if the allocation he makes is the optimum one that maximises the benefits to society.

Here economics has a very useful role to play. Instruments of economic analysis make it possible to express the pursuit of each objective in terms of opportunity costs - that is, in terms of the sacrifices of other objectives that it entails. In this way, it is possible to choose that combination of objectives that will provide society with the greatest benefits at least cost. It would be worthwhile, in view of the mounting pressures on water allocation that are already being felt in some catchments, to launch a programme of analysis to ensure that this is achieved.

2.3 HYBRID WATER ALLOCATION APPROACHES : WHEN IS MARKET INTERFERENCE JUSTIFIED?

In a pure free market system, water would be allocated according to the price which users would be prepared to pay for it, which in turn would be determined in part by what the water was used for. Thus, water needed to support human life would probably be the most expensive, although it might be purchased in quite small amounts. Water for irrigation would be priced according to the value of the crops produced, assuming other user sectors (e.g. domestic and industrial) did not out-bid the agricultural sector for the limited amount of water available.

It is often argued that water is somehow "special", with properties that make the market mechanism an undesirable, even immoral, vehicle for its allocation. Usually this argument is based on the fact that water in some uses is a "merit good" that satisfies basic human needs. A moment's reflection reveals the argument to be less convincing than it may seem at first glance.

Firstly, at a rather superficial level, it may be countered that there are other basic needs whose satisfaction is generally left to the mercy of the market, food being the most obvious example. Following this line of reasoning, it is easy to conclude that the allocation of water outside the market mechanism is more a matter of historical institutional development than of any explicit or even implicit pursuit of "morality" by society. But this argument is something of a red herring anyway.

Much more compelling is the notion that it would be possible to allocate only some water by way of a market - it is not necessary to deal with all water in the same way. More specifically, one could continue to allocate water for basic human needs in the same way as at present, while creating a market for all water used for commercial purposes. In this way the efficiency of market forces could be exploited without any fear of inequity or "immorality".

Alternatively, water for commercial purposes could be allocated in the pattern that would have resulted from market competition, without an actual price being charged for it. The point about market forces is that they achieve efficient resource use: the prices that they generate are merely the means to this end. Thus it is possible to achieve the same level of efficiency, without pricing, as long as the quantities that the market would have allocated to each competing user are known.

Following this line of thought, it is possible to use economic principles to simulate a water market in order to determine the ensuing quantities used by various user sectors. This is discussed in Appendix A. The next step, again using economic principles and techniques, is to introduce public sector intervention to reallocate the water, and to measure the gains and losses of efficiency and equity that follow. In this way, it is possible to estimate the benefits and costs that occur when the market is over-ruled by regulation for purposes of water allocation.

3. NON-MARKET WATER PRICING [4]

3.1 ON WHAT BASIS SHOULD THE PRICE OF WATER BE DETERMINED?

3.1.1 Water pricing as a management tool

The pricing of water is possibly one of the most under-used, but potentially most effective, demand management tools available to the water manager. Apart from providing the necessary revenue from which water schemes can be financed, it has the potential to (1)ensure the maximum beneficial use of water, (2)accurately control demand such that the timing of new schemes can be postponed until they are absolutely necessary, (3)curb demand during periods of shortages, and (4)raise revenue (possibly for the subsidisation of water services to the very poor). In South Africa pricing is used primarily to recover scheme costs and, in some instances, to penalise excessive use during droughts. In view of the economic and physical limitations on further water resource development in South Africa, there would seem to be merit in considering the application of demand management strategies such as water pricing more seriously: it could have advantages in terms of reconciling supply and demand and the production of revenue.

More specifically, given the State's urgent need for new revenue sources, pricing could be considered in water-scarce regions, such as the PWV, where continued supply augmentation is extremely expensive. It could be very useful to evaluate the revenue yield, and demand reduction, that could result over the long term from, say, introducing water pricing at modest rates (which would prevent a disruption of the economy over the short term) but with the clearly stated intention of escalating those rates annually. In this respect it should be noted that by using the accepted equivalent basket of goods and services approach as a basis for comparison, South Africa's municipal water Tarrifs are far cheaper than those of many well-watered European nations.

In practice, although pricing water such that the price reflects its correct economic value is rare, fees are levied on water provision in a variety of ways in different

4 This section of the report should be read in conjunction with Appendix A

countries. The main kinds of pricing are discussed below. As is clear from a comparison of these "pricing" approaches with the discussion of pricing provided in section 3.3 of Appendix A, the former have no particular merit from an economic point of view. In economics, prices are the signals that guide the allocation of resources to their most efficient application. To do this, the signals, or prices, must be correct; i.e. they must be a true reflection of relative resource scarcity.

The "pricing" approaches discussed below clearly do not attempt to establish price signals of this nature. Hence they are not supported by any economic rationale. Nevertheless, they do at least serve to establish a price for water, that goes some way towards achieving its economically correct price.

3.1.2 Cost-recovery pricing (CRP)

Cost-recovery pricing, or CRP, for water schemes is widely accepted and applied, both in South Africa and overseas. Its popularity stems from the fact that many public utilities are not permitted to generate profits for providing essential services. CRP is also a basic requirement of many financing institutions as it minimises the risk associated with loans for water infrastructure. However, it is unlikely that such institutions would have any objection to other forms of water pricing so long as the resulting tariff did not fall below the CRP, or that was so high that it affected the viability of the scheme.

CRP can also encourage efficient economic development by discouraging (through high tariffs) water using activities in water scarce areas. Unfortunately, the benefits of CRP are often lost because true costs of water supply schemes are seldom reflected in the water tariff. Also, CRP can result in a water tariff which becomes unrealistically low due to inflation, especially where capital intensive schemes are repaid over a period of sixty years. Such low prices can, and do, undermine the judicious use of water.

Other problems with CRP are that it does not recognise the relative scarcity/abundance of a resource. Its only aim is the recovery of costs over a fixed time period, regardless of the availability of the water. One of the consequences of this is that at the end of a drought some water supply agencies find they have insufficient operating revenue when post-drought water consumption levels do not return to pre-drought levels. This sometimes results in unpopular water tariff increases during post-drought periods when many consumers expect the price of water to decline due to the removal of drought related consumption penalties. Another problem is that in times of abundance, when dams are either full or spilling, CRP does not cater for short term tariff reductions which could encourage increased water use and achieve cost-recovery in a shorter time period.

CRP also encounters problems with phased water schemes or schemes of different ages which become linked as a result of increases in demand. Consumers who have been used to paying a low CRP for water drawn from an old scheme strongly object to sharp tariff increases resulting from the move from scheme-related tariffs to regional tariffs, or the augmentation of the scheme to cater for new consumers.

3.1.3 Production cost pricing (PCP)

Production Cost Pricing (PCP) determines the price of commercial water by estimating its role in the production of goods, and from this calculating what manufacturers could afford to pay for the water and still make a profit. However, if PCP is used to determine water tariffs, there is the likelihood that consumption levels and the subsequent revenue to water supply agencies may be affected. Any price increase of raw materials used in production will decrease the manufacturers profit margin. In order to protect and even restore that profit margin, the manufacturer will attempt to use less of the raw material by using it more efficiently, i.e. less wastage. Whilst PCP, and any resulting increase in water use efficiency, may be desirable from South Africa's overall resource situation, it could cause problems for the cost recovery of certain schemes and the financial viability of the agencies operating those schemes.

Of course if alternative customers can be found for the surplus water then scheme revenues can be maintained. However, existing water rights and allocation agreements are seldom affected by variable consumption levels. There are no 'use it or lose it' clauses in South African water rights. Thus, holders of water rights can retain their rights without either using or paying for the water, and in doing so can prevent the sale of surplus water to other consumers.

PCP can either be applied to a single sector such as agriculture, (although different irrigated crops can have widely differing profit margins) or it can be applied in an aggregate form to a number of different water using industries. However, before aggregated PCP is used in tariff setting it is necessary to first understand the relationship between the price of water and the resulting likely demand for water by each user. Not all multiple water use situations are suited to PCP. For example, where there is a wide variation in the water use and associated profit margins of a number of production processes, some businesses may become non-viable as a result of aggregated PCP, whilst for others the cost of water may remain an insignificant part of the production costs. Therefore aggregated PCP is best suited to those situations where the water use and profit margins of the various production processes are generally similar.

3.1.4 Water scarcity pricing (WSP)

Water scarcity pricing (WSP) is a trial and error approach to limiting demand during periods of shortage by increasing water prices. The purpose is to increase the price of water in excess of the conservation threshold price for each user sector. The conservation threshold price of water is the price at which a particular user or user sector introduces consumption reducing measures as a result of the anticipated cost of water. It may be related to the profit margins described in the previous sections, or it may be a function of ability and willingness to pay. The conservation threshold is usually different for every user sector, and possibly for users within a sector.

The problem with WSP is that it can be an indiscriminate form of water demand control resulting in different impacts from one user to another. Consequently it is often employed in conjunction with 'the more you use the more you pay' sliding scale tariff systems. This approach at least allows for basic or minimum needs to be met without the user incurring financial penalties. Also, the sliding scale need not be uniformly linear or curvilinear, thereby enabling certain users with specific water consumption levels to be targeted for control. In order for WSP to remain justifiable and equitable between user sectors, sector-specific sliding-scale curves may be needed.

3.2 HOW CAN AFFORDABLE TARIFFS BE SET FOR THE VERY POOR WITHOUT UNDER-VALUING THE WATER SUPPLY?

3.2.1 Problem of water supplies to impoverished communities

It is estimated that close on 12 million people in South Africa do not have ready access to a safe and adequate supply of domestic water. The reasons for this are many: overcrowding in underdeveloped rural homeland areas where water is scarce, the failure of water supply infrastructure in developing regions, high utilisation levels from other user sectors, rapid urbanisation, and the slow rate at which basic services are provided to informal settlements in urban and peri-urban areas. Humanitarian considerations, together with national and international health standards, require that water is provided to these communities. However, few such communities are in a position to pay the full cost of such services.

As some of the needs of impoverished Black people are only now being addressed, the unit cost of new services is often in excess of that of similar services which have previously been provided to comparatively wealthy developed communities. The reason for this is that White residents in established municipal areas have benefited over the years from good, cost-effective infrastructure planning and the effects of high inflation on cost recovery based tariffs. In some instances a measure of cross-subsidisation from the industrial to the domestic sector has reduced domestic tariffs even further. These historical benefits, coupled with the ability of White communities to pay for services, tends to accentuate the differences in service provision between Black and White communities. Such a situation can lead to the charge of racial discrimination being levied at infrastructure developers attempting to overcome the backlog of services in Black communities, whilst at the same time trying to achieve the necessary degree of cost-recovery.

Infrastructure developers often respond to such criticism by pointing out that, for services to be sustainable, adequate cost-recovery is essential. They add that the deliberate non-payment of rates for political reasons severely undermines their ability to provide services. Setting aside the obvious political solutions that are required to resolve the problems of service provision to poor communities, what can the economist contribute to the formulation of affordable and seemingly fair, long term water supply tariffs? Due to the physical characteristics of existing water supply infrastructure and the nature of the communities requiring services, this question must be answered separately for urban and rural situations.

3.2.2 Rural areas

Rural communities face a number of problems regarding water supplies which urban communities do not (or should not) have to contend with. For example, in developing a water supply service, attention must be given to the protection of the source from which the water supply is drawn. If it is a ground water supply then the aquifer must not become polluted, nor must over-abstraction occur. Similarly, in the case of spring water, the spring itself must be protected from access by animals and livestock, as well as contamination from storm water runoff. But it is the protection of the flow and quality of rivers and streams which presents the greatest challenge. The water using effects of afforestation and irrigation, and the polluting impact of activities such as mining, can become major threats to rural water supplies if they are not adequately controlled. Clearly the co-operation of other catchment water users, particularly the timber and mining industry, and commercial farmers, is an essential part of rural water supplies. This co-operation can be encouraged and maintained through the introduction of selected economic instruments. Again the role for effective incentives is evident - and to the economist this means appropriate pricing. Pricing in this context could be applied to both limit the exploitation of the resource base to levels that are sustainable over the long term, and to protect it from pollution by waste emissions. It could, in other words, be used to assist in controlling both the quantity and the quality of available water.

Another problem in rural areas is that the ability of communities and individuals to pay for water supply services is very low. Also, because the water supply is visibly drawn from natural sources, often with the minimum of high technology, communities question the reason for paying for a resource that already belongs to them. Thus, the willingness to pay in rural areas is quite low, particularly among male community leaders, who seldom consider the real cost of carrying water over a distance of several kilometres.

These problems are central to the provision of rural water supplies. To resolve them will require firstly a broadening of the community decision-making structure to include women, as it is they who are largely responsible for a community's water supply. Secondly, with some specialist educational support, the rural water issue needs placing in context such that the real benefits of an upgraded water supply (e.g., community health, increased time available to women for crop production and family care, improved potential for economic growth, reduced need for State aid) can all be considered in monetary terms and compared with the cost of upgrading. World-wide, the set of economic techniques broadly described as cost-benefit analysis (CBA) have made a useful contribution in identifying and justifying worthwhile projects. In South Africa, with overwhelming demands being made on public funds by competing types of social infrastructural spending, CBA may have an especially important role to play in ensuring that the many indirect benefits of improved rural water supplies are fully appreciated, so that water provision is not overlooked in favour of other sectors where the benefits are more discernible.

3.2.3 Urban areas

Urban water supply problems differ from those of rural areas in that population densities are much higher, thus economies of scale render the unit cost of water supply much lower. Also, many communities urgently requiring new or improved water supply services are in close proximity to bulk infrastructure which, if planned correctly, should either have surplus capacity or be upgradable. Supplies drawn from such infrastructure usually have a high assurance of supply which is only affected by extremely severe droughts. Unfortunately, these two key factors (economies of scale and existence of bulk infrastructure) do not seem to have alleviated the plight of the township or informal settlement dweller. Their water supply situation is often more desperate than that of the rural inhabitant, and because of the higher population densities, the risk of epidemics from contact with untreated water is much higher.

The reasons for this state of affairs are many. Some have their roots in Apartheid policies. The artificial establishment of townships, the lack of legitimacy of township councils and the recent collapse of many such administrations, the removal of influx control without any workable corresponding urbanisation strategy, the reluctance (and in some cases inability) of provincial and White local authorities to take responsibility for township services, and the seemingly self-defeating rates boycotts designed to demonstrate dissatisfaction and frustration with local authority services. However, it must also be pointed out that many cities around the world have similar service problems without a history of Apartheid.

With such an array of politically orientated problems, the question might be asked as to whether there is a meaningful role for economics in the provision of urban water services? The answer is yes. Economics can make a contribution by ensuring that the maximum benefit is derived from the scarce funds that will be available for new and upgraded services.

Assuming that political conflict will be reduced after democratic elections, there will be the need to expedite urban reconstruction and development in impoverished areas. In view of the historical injustices surrounding service provision, the various options for levels of future services, the increased funding likely to be available to local authorities, and the various ways in which the cost burden of new and upgraded services can be distributed, the task of urban development will be a complex and daunting one with few clear-cut courses of action. The traditional engineering solutions that have been adopted in the face of water quantity and quality problems are insufficient to deal with such complexity. Economic approaches will be needed in identifying cost-effective options for service provision and in maximising the net benefits from such services. It will be necessary to present options in a clear and understandable manner so that elected community representatives can take informed decisions. It will also be necessary not just to identify the good options, but to select the best options. With their ability to capture complex trade-offs within a coherent and cohesive framework, economic decision-making techniques are well-suited to this task.

3.2.4 Pressures on the setting of affordable tariffs

The pressures that will be placed on those responsible for the future setting of affordable and seemingly equitable water tariffs will include:

(i) The need to ensure the upgradability of services to meet improvements in living standards;

(ii) Preventing the entrenchment of poverty by avoiding non-upgradable rudimentary services;

(iii) Job creation in the development and maintenance of infrastructure;

(iv) Accommodating the demand for different levels of service by different customers within a community;

(v) Introducing the concept of service charges to those communities that have never had to pay for services before, particularly in situations where the current level of service is high;

(vi) Dissatisfaction in communities where negotiated flat-rate tariffs (use as much as you like for a fixed payment) have to be abandoned in favour of 'pay as you use' systems to ensure the sustainability of services;

(vii) The need to demonstrate equity in the cost and level of services provided to developed and developing communities in close proximity to each other;

(viii) Provision of basic services to the maximum number of people in the shortest time.

3.3 HOW CAN SUBSIDIES BE USED TO RESOLVE CERTAIN WATER SUPPLY PROBLEMS WITHOUT THEM BEING MISUSED, MISDIRECTED OR RESULTING IN A DECLINE IN WATER USE EFFICIENCY?

3.3.1 Economic philosophy

To the market economist subsidies are bad. They intentionally distort the prices of goods and services that they affect which can lead to market inefficiencies, misuse and abuse of the undervalued goods and services, and an ever-increasing and often unsustainable financial burden being placed on the community paying for the subsidy. The current GATT negotiations are designed to minimise the role of subsidies in international trade because they are viewed as an obstacle to the economic development of new emerging economies, as well as those economies that are more modernised and efficient. An economy which is characterised by excessive direct or indirect subsidies would be considered as having a flawed foundation on which to develop further. In short, market economics views subsidies as a 'free lunch' that has to be paid for by the beneficiary, often with interest, at some future date.

To the welfare economist, subsidies represent an important means by which different levels of government can intervene in an economy to correct the shortcomings of the market. They can be used to correct inequalities in wealth distribution and levels of service, and protect those members of society who are unable to compete adequately in a market economy. They can also be used to cushion the more harsher effects of market fluctuations on society, and can even be used to stimulate economic growth in certain sectors.

So which view is correct? As is probably expected, both are. There are no pure socialist or capitalist economies in the World. Indeed, since the collapse of the USSR politico-economic system and the more gradual demise of Reagan-Thatcherite economics, many of the World's economies are moving towards mixed economic systems at an ever increasing speed. As mixed economies are characterised by selective intervention by the State in order to achieve certain welfare goals, it follows that subsidies in all their various forms, may be widely used. They are certainly already prolific in existing mixed economies, including South Africa.

3.3.2 Success of subsidies

The next question which must be asked is - do subsidies work? Do they achieve the welfare goals that are intended? Before we attempt to answer this perhaps another interesting aspect of subsidies should be examined. That is, if subsidies are considered so bad by free-marketeers why do they not object more strongly to there introduction? Why do wealthy subsidisers not object to subsidising goods and services for other communities? There are two main reasons why objections are seldom raised over the introduction of subsidies.

Firstly, subsidies are sometimes viewed by the wealthy as being a limited, and therefore more acceptable means of distributing wealth. They are certainly preferable to more conventional socialist economic policies such as wealth taxes or the nationalisation of the production of goods and services. For example, in order to preserve its independence, a privately run service utility may consider it prudent to extend subsidised services to those communities that are unable to pay the full cost of the service, rather than risk being forcibly absorbed into the public service for ignoring the needs of disadvantaged, but politically powerful, communities. Such subsidies are usually financed by charging the unsubsidised consumer more for the services received.

The second reason why so few objections are raised over subsidies is that potential objectors can sometimes benefit directly from such subsidies. In some situations there are flaws in the targeting and administration of subsidies which permit those that do not require them the most to legally obtain them. Administrators often make a basic error in trying to prevent the misdirection of subsidies. They invariably try to 'plug' loopholes in the system by making it more difficult for an applicant to obtain a subsidy. However, in doing so they usually make it more difficult, if not impossible for the intended subsidy recipients to obtain the assistance they so badly need.

For example, the one-third capital subsidy offered by the Department of Water Affairs and Forestry on new or improved water care works was introduced when effluent standards and effluent discharge policing became more rigorous. Its main purpose was to assist municipalities with limited resources in achieving the required level of water and waste water treatment. This is a very common type of subsidy offered by governments throughout the World. However, in South Africa it was originally introduced with unsophisticated mechanisms for separating deserving applicants from less deserving applicants. As a result large municipalities, whose ratepayers could often afford the full cost of upgraded water works, obtained the subsidy and in doing so made a significant drain on the subsidy budget as their works were often quite large. The Department was quick to limit the subsidy to smaller, but not necessarily poorer municipalities.

This particular subsidy system was not originally intended to be used for impoverished Black municipalities, although such municipalities in the RSA are now able to apply. However, many find it difficult to make use of such a subsidy. In the first place, they are seldom able to acquire the finance for the other two thirds of the capital costs. Furthermore, if they obtain a grant from another source which contributes to the other two thirds of the capital cost, its value may be subtracted from the Departmental subsidy. Secondly, few Black municipalities can afford to appoint consultants to draw-up proposed designs and prepare the accompanying application for a subsidy. Although technical and financial assistance may be available from the Department for this purpose, this assistance is usually dependent upon the envisaged works being of a certain 'First World' standard, or achieving a certain quality standard in water and wastewater treatment. As a result, 'sub-standard' or rudimentary water works can be excluded from the subsidy despite the fact that they may considerably improve a community's living conditions or reduce the pollution of rivers. Thirdly, many disadvantaged communities in rural and peri-urban areas do not qualify for a subsidy because no formal local authority exists to represent them.

The above example demonstrates that even with the best intentions subsidies can be misdirected. It also shows that once a subsidy is focused it can be difficult to re-orientate it to a different target group without significant restructuring.

3.3.3 Water subsidies

Subsidies come in two general forms: direct and indirect. Both types are used in the setting of water supply tariffs in South Africa. Direct subsidies can be obtained on the capital cost of water works (see 3.3.2) and water supply tariffs where the consumer cannot afford the full cost-recovery tariff. Indirect subsidies can take the form of technical inputs (planning, design and construction) provided by the authorities but not reflected in the costs of a water project, the writing-off of debts, and the transfer of infrastructure to consumer organisations at less than the market price.

The bulk of South Africa's water subsidies are in the form of reduced water supply tariffs. Most of these are targeted at irrigation water which, if charged at its full cost recovery price, would bring an end to the irrigation of all but the highest value crops in South Africa. However, there are an increasing number of poor rural and urban communities which receive basic water supplies subsidised either by the State, Provincial Government or aid agencies. South Africa's most subsidised (by the State) urban water supply scheme is probably the desalinated groundwater supply to the remote communities of Bitterfontien and Nuwerus in the North-western Cape where consumers pay between 50 cents R1 per kl of treated water which costs over R14 per kl to produce. However, this price is only manitained for basic consumption levels as a 'more you use - the more you pay' sliding scale is applied in this instance. Indirect water subsidies are also common in South Africa although because of their insidious nature it is difficult to determine either their total cost or the degree to which they distort the price of water. The majority of water subsidies are paid for from central treasury funds.

Setting aside the merits of water subsidies, which is dealt with in the next section, it is generally considered that subsidies should be structured such that they do not encourage either the abuse or misuse of water. In view of this risk, it is important that subsidies are regularly and openly reviewed to assess the need for either their continuance or adjustment. However, this review can be difficult if conducted on a non-homogeneous sectoral or community basis

Reviewing indirect subsidies to determine their success or efficiency is an extremely difficult task and is beyond the scope of this study. Such subsidies are often the result of political or administrative decisions. They probably possess limited welfare or development merit and serve only to distort water prices to an unknown degree. It is unlikely that they result in the misuse or abuse of water, although it is possible that certain recipients have become accustomed to such subsidies. Ideally such subsidies should be removed or openly converted to a direct subsidy.

3.3.4 Role of economics in the use of subsidies

Subsidies should be judged not on their size nor on the recipient, but on whether or not they achieve their objective and how efficiently they do this.

Subsidies can be considered as payments (on the current account, i.e., a part of the National Accounting Procedure) given by central government and local authorities to enterprises in both the public and private sectors. They are in effect DIRECT PAYMENTS to recipients. Cross-subsidies on the other hand are contributions made by one group which benefits another group. A current form of subsidy in South Africa is the money provided for housing for the poor. Such subsidies could comprise reduced rent or mortgage interest. However, it is quite easy to show (using economic welfare theory) that income subsidies are superior welfare instruments to rental subsidies.

With respect to water development projects, the choice of either 'broad area' or 'highly localised' subsidies depends upon the type of regional development contemplated. The polarisation of different regions and/or parts of a region is an important consideration as to which form of subsidy will be the most efficient. If investment incentives are restricted to particular growth centres, then localised subsidies may be difficult to implement since concerns with other parts of the region's welfare gains have to be examined.

Many sorts of subsidies fall under the heading of tax allowances. In particular accelerated depreciation of an investor's assets. Such subsidies could be important for private water supply activities in remote regions. The depreciation would encompass such things as plant (water treatment installations) and vehicles etc. A disadvantage that a supplier of water in a poor area would face under these circumstances is that such subsidies depend upon him making a profit. Where no profits are made there can be no subsidies. As a consequence attracting water supply capital to poor areas as the basis for such subsidies is fraught with difficulties. Under these conditions new entrants to the business will have early years of operation where profits are zero and even losses occur. This suggests that for maximum response in attracting private capital for water supply in remote regions, "starting-up" and "running-in" subsidies would have to comprise cash grants and training assistance.

In essence, most subsidies are really tax reductions of an operating cost nature. For example, a frequently encountered subsidy is the PAYROLL SUBSIDY, or wage subsidy. Such an instrument can be used as part of a regional development policy and has obvious "job creating" advantages. Here under-utilised labour factors are used in preference to capital factors. One of the advantages of a payroll subsidy is that it takes minimum amounts of information to operate and bureaucratic interference can be minimised. Such a subsidy could be used as an instrument for encouraging private enterprise to consider water supply business opportunities in the unserved areas of South Africa.

The phenomenon of subsidies and cross subsidies has attracted considerable attention in recent years, and will become a focal point of policy when a new government takes power in South Africa. It is well to know that the effect of subsidies on the allocation of resources (which includes water) is considered to be detrimental by many. Cross-subsidisation in particular has been connected with the distortion of the price-mechanism for commodities and services, as discussed previously.

In the economic literature the concepts of direct subsidies, indirect subsidies (more will be said of these in section 5.1.4) and cross-subsidisation have not been thoroughly examined, and much debate continues. It seems that considerable empirical research is still required concerning the types of cross-subsidisation and their connections with the output from existing investments, and potential investments in projects such as rural water supply schemes. It is important to distinguish between cross-subsidisation and price discrimination. Price discrimination may provide unambiguous welfare gains whilst the gains from cross-subsidisation are not so easily analysed and may, in certain circumstances, induce welfare losses to some members of society. However, cross-subsidisation has been used effectively to ease the problems of transitions in developing economies and this may be its greatest appeal in the new South Africa. They do therefore have a role to play as far as economic instruments are concerned and should not be dismissed as being all bad. Ideally, each case must be looked at on its own merits and not encompassed within general approach.

It has been proposed by some economists that direct subsidies should supplement cross-subsidies. This would involve identifying welfare activities on which direct subsidies could be focussed while all other allocation decisions could be made by market mechanisms. This is a worthwhile possibility for the provision of water in South Africa.

In defending subsidies and cross-subsidies from the criticisms of those who believe the "invisible hand" of the free market can do no wrong, it should be noted that their existence does not necessarily result in resource (water) misallocation. It may even show the opposite. Moreover, the practical application of the market mechanism is less effective than is generally thought in providing an efficient allocation of resources. For example, certain important criteria such as, marginal cost pricing, free competition and perfect knowledge which ensure a free-market mechanism, seldom exist concomitantly in the real world.

4. WATER POLLUTION [5]

4.1 WHY DO WE POLLUTE OUR RIVERS AND STREAMS?

There are two basic economic reasons why we pollute our water courses. Firstly, the effluent which is discharged into rivers is believed to have no value, and secondly, the aquatic environment into which it is discharged is also believed to have no value. These are erroneous beliefs as they stem from incorrect and inadequate pricing mechanisms which prevail as a result of the current policy and legislation governing the use and protection of our natural resources.

If water is priced relatively cheaply, the tendency will be for a user to discard the used water and buy more clean water. Also, if there are no incentives to minimise the degree to which the discarded water is polluted (as opposed to just meeting uniform standards), then it is inevitable that the receiving water will gradually become more degraded. Although most water users pay a volume related tariff, there is no similar charge levied for using the aquatic environment for waste disposal. The waste water discharged by the polluter does have a value, but existing systems do not reimburse him for returning that water to the channel of origin.

Similarly, sometimes the pollutant load or a part thereof, if recovered, can be re-used in the production process. Unfortunately, the price of raw materials used in production (other than water) is not always sufficient to encourage their recovery from the waste stream. However, if a charge were to be levied on the polluter for using the aquatic environment for waste disposal, then this may provide the additional financial incentive for him to introduce pollutant load recovery and re-use. In short, waste water is under-valued because it is not incorporated into existing water pricing systems, and there is no incentive to consider the value of the associated pollutant load.

The value of the effluent receiving environment is similarly ignored when effluent disposal is considered. Although the aquatic environment has a very real, and often high value, to those who make use of it and depend upon it, this value is seldom

5 This section of the report should be read in conjunction with Appendix B.

expressed in monetary terms by pollution regulators. Consequently, the decrease in the value of the aquatic environment, as a result of effluent discharges, remains unknown. Sometimes downstream water users whose economic activity is negatively impacted by upstream pollution can quantify the damage incurred and make a case out for stricter upstream emission control. However, such impacts are not so readily quantified for in-stream water uses. It is not that society does not value the aquatic environment, but that this value is not expressed in a form which regulators can understand and respond to.

4.2 HOW MUCH SHOULD THE POLLUTER PAY?

It is the policy of the Department of Water Affairs and Forestry that the polluter should pay for the abatement of his own pollution. This policy has been adopted by many governments throughout the World. In principle, as is explained in Appendix B, it is possible to calculate exactly how much a polluter should pay to fully internalise the negative externalities imposed on the rest of society. In practice, making this calculation is very difficult, largely because the benefits of reducing pollution cannot be estimated precisely. Usually the pragmatic solution chosen for this problem entails relinquishing a pure economic approach in favour of a standards and charges approach, in which the degree of pollution abatement required is set by reference to society's preferences (instead of on the basis of calculated economic costs and benefits) and charges are then fixed at the level required to achieve that standard.

4.3 IS SELF-REGULATORY POLLUTION CONTROL FEASIBLE?

It is shown in Appendix B that one of the advantages of the economic approach to pollution control is that it allows for a degree of self-regulation by industry. The extent to which this can be achieved will depend mainly on the physical and economic characteristics of a particular geographic area. Where these are such as to create an ideal "bubble" within which to introduce "tradable permits" (see Appendix B for a discussion of these terms), the government may need to do no more than determine an overall emission standard or load for the area and the penalties payable if this is exceeded, and then monitor its observance. Beyond this, industry itself can be left to devise the necessary institutions and incentives to implement the standard amongst its various members.

the necessary standard. The subsidy can be administered in the form of a capital grant., However many countries now find it more efficient to administer it in the form of tax concessions to companies investing in pollution control equipment. Such a concession is obviously dependent upon emission standards being met and, in some countries, the loss in revenue being off-set by income from pollution taxes. Another form of this type of subsidy, which was adopted in South Africa, at one time, is for the State to fund research and development initiatives which assist industry in cost-effectively reducing its pollution to meet the required standard. The past activities of the Council for Scientific and Industrial Research in assisting industry in this regard, are well documented.

Such cases are probably quite rare, however, and in the majority of situations less self-regulation will be possible and a concomitantly greater degree of government intervention necessary.

It should always be recognised however that the range and type of skills present in, or available to, effluent producing industry are often capable of achieving a high level of waste management and control providing the correct incentives are in place to enlist their services in view of South Africa's skills shortage, particularly in the public sector, this is a resource which we cannot afford to disregard.

4.4 SHOULD GOVERNMENT TAX POLLUTION RATHER THAN PROFITS?

The main taxes universally favoured by government - those on personal or company income - have a generally negative effect on productivity. That is, they have a disincentive effect on something that one would normally wish to encourage. By contrast, "green" taxes have a disincentive effect on something that is to be discouraged, namely pollution. From this point of view, pollution taxes may be attractive as a component of the national tax base.

However, it must immediately be recognised that pollution *per se* does not meet all the criteria of a good tax base. In particular, it may be self-liquidating, and a dilemma is encountered here which has not been dealt with satisfactorily by many countries to date.

There are two choices when it comes to levying a pollution tax. The first is to do so at a modest rate without major disincentive effects; this will provide an ongoing source of revenue, but not eliminate the pollution. The second option will penalise pollution more severely, serving to reduce and perhaps ultimately eliminate it, and at the same time removing the tax base itself. A pollution tax that is attractive to the fiscus will therefore not meet the aims of environmentalists, and vice versa. However, in South Africa there may be considerable advantages to be had from the latter at this particular time. A pollution tax that was self-liquidating over, say, a decade, and both generated revenues and re-directed economic growth away from polluting activities, could play a useful role during the phase of economic restruction that the country is about to enter.

Both forms of pollution tax are used extensively and successfully overseas, although as might be anticipated, the low level, revenue generating tax is by far the more popular. It should be noted that pollution taxes do not always find their way into the general fiscus of a nation. In many instances the revenue generated is earmarked for the funding public sector environmental control activities, which often results in the more rigorous implementation of enviromental policy as the responsible agencies are automatically less vulnerable to cut-backs in State expenditure.

The disbursement of revenue from pollution taxes is discussed further in the following section.

4.5 PAYING POLLUTERS NOT TO POLLUTE

A frequent motivation for the introduction or an increase in pollution taxes, used by many governments, is the need to fund subsidy programmes for pollution reduction and abatement equipment. The critics of of such programmes often refer to this as 'paying the polluter not to pollute', and claim that it is contrary to the "Polluter Pays Principal'. Whilst this is technically the case, such programmes must be considered in the broader context of economic efficiency, and, where poorer nation are concerned, welfare benefits. This latter aspect is dealt with more fully in section 5.2 which discusses sanitation costs and subsidies, while this section focuses on the issue of State subsidies for pollution control expenditure by industry.

One of the main drawbacks of the use of rigid emission standards under a Command and Control system, is that they tend to be indiscriminate towards polluters. In other words, standards focus on pollutant emissions and completely ignore the socio-economic circumstances surrounding such emissions. Consequences such as reduced competitiveness, factory closure (particularly older plants) and loss of jobs etc. are not accounted for within the Command and Control system.

The response of South African water managers to this situation has been to issue permits which exempt the polluter from complying with certain aspects of effluent discharge regulations. Whilst these permits are periodically reviewed, they are seen by many environmentalists as a 'licence to pollute'. Furthermore, as the circumstances surrounding the granting of individual permits are considered to be confidential by the authorities, environmental groups maintain that they are not given the opportunity to adequately consider and challenge the motivation of exemption permit applicants, particularly the economic arguments presented.

An alternative to exemption permits is to make subsidies available to polluting firms to off-set the cost of installing new emission control equipment capable of meeting

5. WATER INFRASTRUCTURE DEVELOPMENT [6]

5.1 WHAT ARE THE REAL COSTS AND BENEFITS OF A NEW WATER SUPPLY?

5.1.1 Understanding full cost-recovery

'Full cost recovery' is a term which is often used to describe a tariff structure and demand projection which ensures the repayment, in full, of the capital, interest, operation and maintenance costs from the consumer within the expected lifespan of a water scheme. In other words, the money which is required to build and operate the scheme is recovered from the consumer by means of tariffs. However, this is a long way from the full economic cost recovery of a scheme.

The full economic costs of a scheme generally includes three additional types of costs: impact costs not included in the cost of the scheme, opportunity costs resulting from a shift in natural resource use, and hidden public sector costs. These costs and their estimation are described more fully below. The unaccounted impact costs of a scheme invariably comprise social and environmental costs.

5.1.2 Impact costs

(a) Social impact costs

Some of the social impact costs of a water scheme should be included at the planning stage of a water scheme. These costs are usually based on the direct costs incurred by individuals or communities as a result of a scheme. This normally entails agreed compensation for land inundated by impounded water, or servitude payments for pipelines and canals which cross private property. However, what is seldom assessed and compensated is the impact of water schemes on rural community structures, lifestyles and traditions.

The impact of the construction process alone is acknowledged by planners and administrators to be immense. So much so that it often fuels debate as to whether it is more beneficial to involve local labour and entrepreneurs in the building of a

6 This section of the report should be read in conjunction with Appendices A and B.

scheme, knowing that the overheating of the local economy will be short-lived and painfully terminated, or whether it is better to protect the community from such upheaval by locally importing all labour, supplies and services and isolating the community from the construction process. Whilst there may be sound sociological arguments for the latter, the former approach is often applied.

Part of the reason why water scheme constructors impact so greatly on rural communities is the poor level of institutional capacity present in many such communities to represent fully the interests of the local people. Many developers believe that it is preferable to negotiate numerous compensation agreements with individual land owners, especially if rural economies are depressed, than to have to negotiate a single agreement with a cohesive representative body who are determined to achieve the best compensation deal possible. The difference between the agreed social compensation costs of a water scheme and the real social impact costs, is an externality which has only recently been acknowledged in South Africa. In trying to minimise the financial cost of a scheme, and thus the unit cost of the water supplied from it, planners risk maximising the social externalities, particularly in situations where institutional capacity is low and rural poverty is severe. As all externalities, both social and environmental, represent accumulated debt which has to be paid at some future date often with interest, there is little economic gain in minimising the financial costs of a scheme by increasing the externalities.

One type of social impact of water schemes which is often overlooked is the effect on local communication of new impoundments and rivers with artificially high flows. Also the river flow impacts of upstream abstraction and river regulation on downstream communities, their water supplies, their cultivation of the flood plain, and their supply of fish, has only started to receive attention in the last few years.

A good example of the post-development internalisation of social externalities of a water scheme in South Africa is the development of the management plan for the Phongola Floodplain downstream of the Phongolapoort Dam. In this particular instance a dam was built and an operational release policy was followed which did not initially consider the needs of the indigenous people residing near to, and making a living from, the floodplain. The plight of these communities was addressed in earnest following severe floods in the 1980s which devastated the floodplain causing widespread hardship to all that depended on it.

(b) Environmental impact costs

Environmental impacts are mitigated to a predetermined degree in all development projects today. This degree is often arrived at by means of Environmental Impact Assessments and Integrated Environmental Management procedures. However, both these techniques endeavour to find compromises between development and conservation, particularly where the future supply of an essential commodity such as water is concerned. If the natural environment that is to be negatively impacted by a water scheme is not ecologically important, or is not situated in a nature reserve or an area recognised for its natural assets and beauty, then the chances are that it will be awarded a low value which will subsequently reduce the estimated environmental impact costs.

A limitation of current techniques for valuing the environment is that they tend to weight the valuation in terms of species composition (scarcity and biodiversity) and utility to the present generation. The geographic distribution of common natural assets or their potential value to future generations are seldom considered.

For example, supply and demand mechanisms promote the high valuation of a natural asset only once man has destroyed most other examples of it. The fact that we place such a high value on threatened species and biodiversity today is because development has gone unchecked in the past. However, conservationists would be quick to point out that this is similar to saying that to preserve natural assets one must first render them scarce. This has certainly been the case with South Africa's dwindling wetlands. Wetlands, and in particular vleis, are one of the most sensitive types of natural habitat to water resource development. Apart from the fact that many vleis are excellent potential dam sites, their dependence on specific river flow regimes renders them highly vulnerable to upstream abstractions and river regulation. The rapid water resource development of the last 35 years has taken a heavy toll on vleis, many having been dried-out, drained or impounded. This has had the effect of increasing the value that society attaches to the remaining vleis and wetlands to the extent that it would be very difficult today, if not impossible, for a recognised wetland to be destroyed by development.

Similarly, if society does not make use of a natural asset, or is not aware of its existence then there is a risk of its undervaluation, regardless of whether or not it is ecologically important. Conservationists who have diligently protected important natural assets from human impact by preventing public access to them, often have difficulty in understanding why decision-makers and society place so little value on these assets. A recent example of this was the proposal to build another dam in the

Kogelberg for future water supplies to Cape Town. Although the dam would have inundated an extremely valuable and internationally recognised area of Cape flora, the response of the intended recipients of the water (people living on the Cape Flats) to the impact was, to say the least, ambivalent. Having been prevented from visiting the site for so many years many people had little appreciation for its beauty and plant diversity, and thus only saw it as the cheapest source of water. This situation prompted conservationists to develop proposals for nature trails and educational visits, in a bid to save the area from flooding.

(c) Potential inputs from economists

Clearly we have to stop viewing the social and environmental externalities stemming from a water scheme as a debt that does not have to be repaid. These externalities are economic liabilities which will have to be internalised, to a socially acceptable degree, at some stage in the future. This may occur as a result of public and media pressure, awareness of the welfare and environmental obligations of the State, or even litigation. The cost of this internalisation may fall to the water consumers, the tax payer, or alternatively a future generation. Economic theory and the polluter pays principle states that the water consumer should pay as soon as possible to avoid the inefficiencies associated with State subsidies (the tax payer) and intergenerational debt.

Economics can assist in internalising externalities associated with a water scheme by :

- identifying and describing them,
- quantifying them (sometimes, however, this is very difficult to do),
- alluding to the consequences of ignoring them, and
- calculating the degree to which they should be internalised to ensure optimality and sustainable development.

The techniques that can be used for this are described in Appendix C.

5.1.3 Opportunity costs

The opportunity cost of water is the income that is lost when water is allocated to a use which is unpriced or non-market related, in preference to a use which directly generates wealth. For example, the opportunity cost of allocating water for the conservation of a natural area in preference to an irrigator, would be equivalent to the additional income the irrigator could generate if the water had been allocated to him.

Opportunity costs are used to assist water allocation decision-making by indicating the relative trade-offs between the commercial and non commercial uses of water In certain situations they can be used to optimise the internalisation of the external costs of excessive river abstractions

5.1.4 Hidden public sector costs

Identifying and exposing hidden public sector costs is important since the majority of water resource developments involve public sector organisations which are funded from sources other than the sale of water. The likelihood of this is that some of the cost of developing and operating a scheme may not be reflected in either the overall budget or the cost of the water delivered.

For example, it is the responsibility of the Department of Water Affairs and Forestry to undertake the planning of bulk water supply infrastructure. This is an expensive undertaking which costs the tax payer many millions of Rands each year. Individual planning studies can take several years to complete and involve numerous consultants in a wide variety of fields, at considerable expense. These costs are not added to the cost of the water scheme and are not recovered from the water consumer except via the indirect route of individual and company taxation. In other words the exclusion of planning expenses from the cost of a scheme is an indirect State subsidy which artificially suppresses the unit cost of the water supply.

In other countries where the costs of planning a scheme are reflected in the scheme budget, planning costs can amount to between 10-20% of the total capital costs. The main risk with planning water developments from Treasury funds is that it is highly vulnerable to cutbacks in State expenditure, particularly if the perceived urgency of a scheme has been diminished by several years of good rainfall. Indeed a risk situation could arise where a viable proposed water scheme with a high potential for full financial cost recovery (thereby making capital borrowing relatively easy) is delayed because of the under-resourcing of the State's water supply planning function.

Once a scheme has been planned and designed, and the various impact studies completed, the Department may suggest that it constructs and operates the scheme using labour and equipment which is again funded from Treasury and not recovered from the water consumer. In this respect the Department has a proud history of constructing and operating schemes to an internationally recognised high standard. However, this reputation did not dissuade the private sector from strongly objecting to the Department's construction activities. The outcome of this objection has been :

- the rapid reduction in the Department's Directorate of Construction,
- a requirement for the Department to show all its costs associated with the construction of a project, and
- the building of large water schemes by the private sector.

The operation of water schemes by the Department has also been a source of hidden costs over the years, partly due to labour and equipment which was paid for by the tax payer, and partly due to the absence of proper trading accounts for the individual schemes. Although the Department regularly adjusted the water tariff to urban consumers to cover increasing operational costs of water treatment, few such increases were passed on to irrigators as most of the operational costs associated with their supply were in the form of publicaly funded labour and equipment.

Once again this type of subsidy has begun to be phased-out in recent years by means of the transfer of schemes to local agencies such as irrigation boards. Such agencies are supposed to maintain and operate schemes, and to recover the costs of this from the consumer.

The hidden costs, or indirect subsidies (cf. section 3.3.4), introduced by the Department to many water supply schemes, were originally intended in the early part of this Century to open-up the interior of the country to farmers: such subsidies were also a feature of opening up the American West. This system was used to great effect in the 1960s and 1970s, in conjunction with direct State subsidies, to supply irrigators with artificially cheap water. Today, the cost of this externality is paid for by all South Africans in the form of the degradation of the nation's natural resources due to ill-considered and unviable irrigation projects.

The solution to hidden costs and associated water price distortions is a straight forward one, - <u>all</u> the costs of supplying water should be reflected in the accounts of a scheme. However, this creates two problems namely: where does the money come from to plan and design water schemes?, and what about those consumers that genuinely require State aid in developing a water supply scheme?

In the first instance if the projected demand for water exceeds the available supply, and the willingness and ability to pay for an augmented supply exceeds the anticipated costs of a possible scheme, then there is no reason why the funds required for the planning and design of a scheme should not be borrowed on the capital markets. This loan could be serviced by the State with the full planning and design costs being added to the scheme budget at a later date. Alternatively, where a paying consumer base is already well established, the planning and design loan could be serviced by a levy on the existing water tariffs. This to some extent has already been done in the PWV area to help fund the development of the Lesotho Highlands Water Project.

In the second instance, i.e. those consumers that cannot afford the full costs of the planning, design and construction of a water scheme, the same principles of full open accounting should apply, the difference being that the State must assume responsibility for the repayment of any loans. It is generally considered that the minimum contribution that poor people should make towards their water supply costs is to cover the operation and maintenance costs. This way the financial sustainability of the scheme is ensured.

5.1.5 Unaccounted benefits

(a) Background

The previous sections have considered at length the costs associated with water schemes. It is important to balance these costs by examining the corresponding benefits that arise from investments in water schemes, especially those that are often poorly articulated during the motivation and planning of a scheme. It should be noted that such benefits are difficult to quantify and even more difficult to attribute exclusively to the provision of water. The following notes examine some of the benefits that are often not taken account of in calculating the benefits that accrue from a water supply investment.

(b) Role of water supply in development

A common misconception is that the provision of a water supply to an area can stimulate economic growth. This belief has arisen out of various economic studies which have examined the reasons why certain areas have not developed as well, or as fast, as other similar areas. As most of South Africa has a semi-arid climate, and because the conditions required for economic growth are highly complex and not always well understood by analysts, such studies tended to identify an inadequate water supply as being the main culprit for economic underdevelopment.

The fact is that a water supply *per se* cannot stimulate economic development. The provision of new or upgraded water supply infrastructure can support an economic development trend and help it maintain momentum. Similarly, the failure to meet the demand for water timeously can slow the rate of economic growth, although this is a debatable issue in view of the relatively low price of water and the associated high degree of wastage, occurring in South Africa. Many economists would suggest that a water shortage situation is an ideal opportunity to correct prices and improve water use efficiency. They would probably add that the automatic provision of additional water to meet rising demand without questioning consumption patterns, is an uneconomic approach to water management.

There are several examples in South Africa of where elaborate water infrastructure has been provided but the anticipated economic development failed to materialise. The Berg River - Saldanna Government Water Scheme is a prime example. It was planned and built at a time when Saldanna was believed to be about to experience an upsurge in economic activity and population growth due mainly to plans for the export of iron ore from the port. Then the World demand for iron ore fell, the economic growth at Saldanna never occurred, and the State was left with an extremely under-utilised urban water delivery system with all the associated operational problems and costs.

Interestingly the port of Saldanna is today operating at a very high level of its capacity. However, the export facilities are highly mechanised and provide limited economic spin-offs for the town, hence the water supply infrastructure remains under-utilised. The attraction of Cape Town as an economic growth point is clearly too strong to permit industrial activity at Saldanna, despite its export facilities.

(c) Assessing legitimate demand projections

The above example demonstrates that water planners, (in view of their responsibility to the nation in managing a scarce resource) must be prepared to closely examine the legitimacy of all demands for water. It is not sufficient to acknowledge that a farmer does indeed own land which can be irrigated if water is made available. The market potential for the crops to be produced, the viability of the production process, and most importantly the real value of the crops (as opposed to the subsidised price paid to the farmer) must be assessed in order to determine if meeting the demand for water is in the best interests of the nation. If disputes arise over economic growth and water demand projections then these can be easily resolved by placing the burden of risk on the advocates of a new water supply. This can be done by agreeing on a non consumption related financial cost recovery tariff for the proposed scheme.

(d) Multiplier effects of water schemes

Investment in water schemes have, like other forms of investment, multiplier effects, both direct and indirect. For example irrigation schemes provide job opportunities on the schemes themselves and indirect job opportunities to the process industry that packages agricultural products, the transport industry that delivers such products to market and the wholesale and retail outlets that distribute the products. The same holds for the provision of water to industrial production processes. Such multiplier effects influence the main macro-economic variables of an economy such as the Gross Geographic Product (GGP), The Gross National Product (GNP) and the Balance of Payments. These effects can be estimated by economists from established Input-Output (IO) tables. Although national IO tables do exist for South Africa, the validity of certain regional IO tables has been compromised to some degree by the lack of data emerging from, and the distorted economics of, the homelands

More recently, IO tables have been augmented by Social Accounting Matricies (SAMs) which better reflect the employment opportunities stemming from infrastructure investments. Unfortunately, neither existing IO tables or SAMs fully reflect informal economic multiplier effects of sectoral investments. In particular, the multiplier effects of new water supply schemes to disadvantaged rural communities are almost impossible to estimate at present as the economics of subsistance agriculture have yet to be studied, qualified and incorporated into economic data bases.

5.2 SANITATION COSTS AND SUBSIDIES

The access of people in South Africa to proper sanitation facilities is far worse than the water supply situation. Unfortunately, sanitation is not seen as a particularly high priority in many quarters. While a safe water supply is essential for survival, inadequate sanitation is considered by some to be just an inconvenience. Although the official policy of local, regional and Government departments is to ensure adequate sanitation, there is little evidence of it in rural areas. Even the recent outbreak of typhoid in Delmas (probably attributable to poor sanitation) has done little to change existing policies, despite being accompanied by calls for more funds for sanitation. The fact is that, when undertaken in a top-down manner, the provision of sanitation in rural areas is an expense that no administration wants to consider.

There is a range of sanitation systems available to rural communities, from soak-away systems such as pit latrines, septic tanks and French drains, to more costly small bore, solids-free water-borne sewerage systems. For each system there are a number of variations to suit both the socio-economic status of a community and its longer term aspirations.

Although low cost systems are usually selected for the provision of basic sanitation service in rural areas, such systems may not always be suited to the physical site characteristics. Also, as a community grows, the original sanitation system may become inadequate or unsuitable. In the case of soak-away systems, the following factors have to be considered:

- Depth, absorptivity and moisture retention capacity of the substrate;
- Slope of the land;
- Proximity to surface and ground water resources;
- Utilisation of the surface and ground water resources which may be affected, especially the protection of the community's water supply;
- Present and future dwelling density.

Where communities rely on ground water abstracted from beneath their site, the likely impact of soak away systems must be investigated in detail to prevent contamination of the water supply.

Moreover, soak-away sanitation systems cannot always be considered permanent solutions for high density situations. As the population grows the filtering and nutrient retention potential of the substrate can diminish until it eventually becomes saturated and ceases to be a safe method of waste disposal. Once a soak-away system becomes inadequate, the choice of alternatives is limited.

Even though rural sanitation options are mainly low cost and low technology based (i.e. community labour can be used), the large number of people in need of them, and their inability to pay the capital costs, means that the provision of adequate sanitation to everyone is a cost burden of enormous proportions. This raises the issue of who should pay for such services and how?

From an economic point of view the answer is straight-forward, the beneficiaries of improved sanitation should pay. The problem is that the beneficiaries have first to be identified and then made aware that they are indeed beneficiaries. For example, the community served by a sanitation system is an obvious beneficiary, but it probably has little idea of the health benefits it is enjoying, or the costs associated with the likelihood of disease if it did not have such facilities. Similarly, the employers and fellow employees of people served by proper sanitation are also unwitting beneficiaries. The environment and the outdoor recreationalists who enjoy it, are beneficiaries of improved rural sanitation in that pollution levels are reduced. Downstream water users (e.g. industry, towns, farms and other rural communities) also benefit from improved rural sanitation either through reduced water treatment costs or through just having a safer water supply.

Despite such an array of potential beneficiaries, the costs associated with rural sanitation remain largely for the account of the community. Consequently, few communities decide collectively to fund sanitation systems. In most cases it is up to individuals in the communities to decide for themselves if they want to install sanitation and how much they want to pay for it. The result of this has been the unco-ordinated and haphazard development of sanitation of various levels of suitability and effectiveness.

The costs associated with sanitation should ideally be distributed according to the benefits enjoyed. In this regard it can be argued that the provision of sanitation reduces the hazard of ill-health, the spread of disease, losses in production etc., and is therefore a candidate for subsidies where the community itself cannot afford the full capital cost of building sewage disposal facilities. The economic rationale is that the lack of sanitation affects the economic well-being of the whole nation and localised subsidies can therefore be justified (cf., section 3.3.4). Consequently, and using cost-benefit analysis techniques, consideration needs to be given to allocating a portion of the costs of rural sanitation among the employers of rural people (timber companies, farmers, tourism industry etc.), environmental groups, and neighbouring towns. In addition, unlike small rural water supply schemes (which often require a small number of highly skilled people to develop, i.e., geo-hydrologists, drillers, fitters and welders, pump and pipe engineers etc.), sanitation schemes have a tremendous capacity for employing unskilled labour within the community. Thus, in addressing the backlog in sanitation services by means of national subsidies, the double-dividend of pollution control and social welfaregains can be achieved.

5.3 SELECTION OF THE DISCOUNT RATE FOR WATER PROJECTS

The determination of the most appropriate discount rate for water projects has been a major issue of debate between financiers, economists and government policy makers throughout the world. It is not uncommon to find planners using a wide range of discount rates with which to calculate the benefits of a single project. Such ranges reflect the uncertainty that surrounds discount rate selection and the hope that someone else (invariably a politician) will choose the rate to be used.

Whereas in financial analysis the interest rate used normally reflects the market rates for capital (including inflationary effects), the discount rate used in economic analysis is not readily apparent from the economy. Hence, economists have developed a number of ways of determining and justifying a discount rate. Three of the most common approaches, the 'Cost of Borrowing Money', the 'Opportunity cost of Capital', and the 'Social Rate', are described below.

Governments frequently have to borrow more, on either domestic or international markets, in order to finance water projects. The servicing costs of such loans can be used to set the discount rate. The problem with this approach is that low or subsidised interest rates will favour projects with long term net benefits, while high rates will favour projects with short-term pay-offs. Another approach is the 'Opportunity Cost of Capital' which is based on the return on investment that the private sector would expect to receive on the money used for a water project. In this case the discount rate reflects the rate of return on capital productivity. This approach is often used by international development banks as it automatically discourages these projects with limited net economic benefits. The 'Social Rate' is based on the ability of society to assess the true long term benefits of a project, which often extend far beyond the loan repayment period. Consequently the Social Rate is invariably lower than the two previously described rates which are more closely associated with the time horizons of the markets.

In summary, the discount rate used in a water project is dependent upon government policy, on borrowing and development, the source of the funds, and more importantly, the beneficiaries of the new supply. Generally, water supply projects with a strong welfare component or a high multiplier effect in a depressed economy, will warrant a low discount rate. Whereas projects for which full financial cost recovery can be expected, or where the purchased water can be used to generate wealth, will qualify for a high discount rate.

THE APPLICATION OF ECONOMICS TO WATER MANAGEMENT IN SOUTH AFRICA

Part 2

APPENDICES

APPENDIX A

WATER QUANTITY MANAGEMENT

- to limit demand; or
- to increase supply.

In the case of a natural resource such as water, the do nothing strategy, which cannot really be classed as a management strategy, will lead to automatic rationing where people will have to queue to receive small quantities of water, or to malpractices such as black markets.

Limiting demand involves persuading consumers that the scarce product (water) is not really desirable or necessary, thus reducing their desire to come by it. In times of drought where water supplies become critical, efforts to limit demand are usually directed towards decreasing non-essential use, and are not overly successful. Shortages in market-governed situations (such as stock markets) are dealt with by the signals sent out by the prices, set by the market as scarcity waxes and wanes. As prices rise the commodity becomes less desirable to the consumer, and thus, the demand falls. Therefore, the solution that an economic approach suggests is to raise the price of those environmental resources which should be conserved.

Increasing supplies is the management strategy usually associated with water management. Water authorities and engineers have traditionally tried to alleviate shortage by making more and better resources available. This strategy is sound enough, but starts to break down as resources approach full utilisation.

In essence, therefore, since the do nothing approach cannot be regarded as a management strategy, the two approaches open to water managers are to manage demand or to manage supply, and the implementation of these strategies is dealt with in the following sections.

CONTENTS

A1. INTRODUCTION

The underlying problem faced by all who would attempt to manage natural resources is quite simply that many, if not all, are becoming scarcer. Dictionaries describe scarcity as a situation where there are insufficient resources to meet the demand under the prevailing circumstances, or an inadequate supply. Both of these descriptions point to a situation where either supply is too small or demand is too great.

Resource managers have effectively four strategies which they might pursue in order to address such scarcity. These are:

- to do nothing;

A2. TRADITIONAL APPROACH

The traditional approach to water management has been to implement a combination of supply augmentation with non-price rationing through regulations. This in effect is supply side management in operation. The rationing side of the approach might be seen as limiting demand, but as it does not limit users' desire for water, but only their access to it, this is really a "policed do nothing" approach.

This report is concerned with the application of economic principles to water management, and this traditional approach does not embody these principles. It is really part of the problem which the use of economic principles sets out to try to remedy. No further time will therefore be spent on this issue. The following sections will deal with demand-side management and water pricing, which embody the economic principles alluded to the above.

Also, there is room for an extension of this system in order to take into account environmental considerations.

Marketable pollution permits, established on the basis of a determined overall level of pollution, can be allotted to individual polluters. Polluters with low costs of abatement control will thus have an incentive to reduce their emission levels and to sell their surplus permits to other polluters. Polluters with high costs of abatement control will have an incentive to buy pollution permits instead of undertaking costly control measures. A pollution permits market can therefore be created.

If the allowable overall level of pollution is determined by the pollution level that the community considers acceptable then the pollution permits will reflect the marginal damage costs imposed on the community in satisfying marginal demands. Thus, damage costs will have been given the appropriate price in order to promote an efficient allocation of resources.

The transferable water rights system is a do-it-yourself pricing system. This is its main advantage compared to the marginal cost pricing system presented above. It relieves water authorities from devising, implementing and administering water charging schemes and, therefore, from the administrative and political costs that such actions may imply. On the other hand market failures may arise with this system.

Monopoly/monopsony situations might occur if there is a dominant buyer or seller as well as externalities due to "public good" effects.

As far as monopoly/monopsony situations are concerned if the market for rights can cover a geographical area including a large number of demanders and suppliers this problem can be reduced.

Externalities due to "public good" effects may be dealt by (a) the potentially damaged party having recourse to administrative or court action, (b) a water authority being responsible for monitoring and perhaps amending potential transfers, or (c) strict geographical specification of areas only within which transfers will be allowed. (OECD, 1989b).

The purpose of the above is not to assert a general superiority of one demand-side technique over another. Its purpose is rather to point out the pricing and transferable water rights techniques.

A3. ECONOMIC APPROACH: DEMAND SIDE MANAGEMENT

A3.1 CONSERVATION — MORAL SUASION

One of the most basic approaches to reducing demand, that is reducing consumers' desire to consume, is to appeal to their moral judgement. This amounts to persuading them that it is morally or sociably not acceptable to continue their present consumption patterns. This approach can be very effective where highly emotional issues are involved, such as the use of animal furs for clothing for the affluent. Campaigns headed by conservation-minded groups have been successful in noticeably reducing demand for such products.

In the case of demand for water, this approach can also provide some effect by appealing to consumers' social responsibilities not to waste such a precious commodity as water. Unless accompanied by some or other punitive measures to underpin the effort, this approach is not generally very productive. Much of the problem seems to stem from consumers' perception that water is a "free" good, God-given to all, and thus to be used according to an individual's perception of his own personal needs.

A3.2 TRANSFERABLE WATER RIGHTS

There is, at present, growing interest in the design of water utilisation rights which may be transferred through voluntary exchange, for example in a water rights market.

The idea is simple: the opportunity to sell or lease a water right provides some incentive to conserve and transfer water to higher-value uses and thus promotes efficient water allocation. Water users for whom water has low use-value will have an incentive to use water economically and to sell or lease their rights for spare water. Water users for whom water has high use-value will have an incentive to lease or buy water rights in order to expand their activities.

Equity issues and protection of the poor may be incorporated into such a system by providing minimum non-transferable water use rights per person, or village, or community.

Political considerations, administrative and institutional constraints, the specific characteristics and problems of every region and the water sector, must be considered. This broader perspective is needed to indicate which method is the most appropriate one, or which combination of both methods could be the best way to promote economic efficiency and environmental effectiveness.

Currently, there is extensive debate and experimentation in most OECD countries relating to the market-oriented techniques compared with that of standards. A brief overview of the debate seems essential.

Transferable water rights are not easily applied to public water supply since the public water supply system involves huge investments in works which are not transferable.

However, if the water authorities responsible for the public supply systems are involved in some transferable water rights system (marketable permits system), for example they could purchase water from agriculture to use in public supply, then they could use the prices inferred by these systems to better calculate their natural resource-depletion (damage) costs.

A3.2.1 Examples of Transferable Water Use Rights for Direct Abstractions

Three examples of transferable water use rights have been identified in the area of direct water abstractions. They cover the United States (Colorado) and Australia (South Australia and Victoria).

"The North Colorado Water Conservancy District has allowed share trading since the 1950s (a "share" is 1/310,000th of the water of the water available to the district each year), and it is known that the changing pattern of water demands in the region has been facilitated by allotment changes (via market transactions) away from agriculture to urban and industrial use (irrigation allotments fell from 85 per cent, 64 per cent of total yield over 1957-82)." (OECD, 1989b).

"In South Australia, water rights policy has been undergoing liberalisation since 1979. By 1984 irrigation, industrial and recreational/environmental surface water allotments in the Murray River basin could all be traded (subject to veto relating to quality deterioration). Over an 18 month period transfers were approved covering 3 900 Ml per annum, about 1.5 per cent of the total quantity licensed for irrigation. A ground water transfer scheme also started in 1984. Land sale analysis suggests that marginal units of water doubled in value following the commencement of the first formal transfer policy in March 1983." (OECD, 1989b).

In Victoria, Australia, the state Rural Water Commission subjected water allocations from river systems to auctions. Auctioning has been preferred to a tender system since the latter prevents interested parties having any information of what others are prepared to pay. The first auction, of 2000 Ml of River Loddon water in May 1988 was highly successful despite strong objections from established irrigators (OECD, 1989b).

A3.3 WATER PRICING - "IDEAL" AN ALL-ENCOMPASSING APPROACH TO WATER MANAGEMENT

A3.3.1 Introduction

Economic theory postulates that an efficient allocation of resources, i.e. an allocation that maximises the community's net benefits, requires that prices reflect the costs to the community of satisfying marginal demands. The principle of marginal cost pricing proposes that, at the margin (for the last unit consumed), prices should reflect the incremental costs imposed on the economy in meeting those demands.

If water authorities price water on the basis of marginal cost pricing, water will be allocated to its highest use-values and a community's welfare will thus be maximised.

A3.3.2 Costs of Water Services

Under marginal cost pricing, prices should reflect all costs to the community of satisfying marginal demands.

These costs may include resource-use costs, natural resource-depletion costs, damage costs or any combination of them.

A3.3.3 Resource-Use Costs

Resource-use costs are those costs associated with the goods or services forgone by the commitment of economic resources to the construction, operation, connection, maintenance, etc. of the supply or disposal system, provided such costs are necessary for the supply or disposal water service.

In fact, the resource-use costs mainly comprise three types of costs (OECD, 1987):

- Customer (or Access) costs
- Commodity (or Volume) costs
- Capacity (or Demand) costs

Customer costs are those costs associated with a customer being connected to a supply or sewage disposal system, even if no water or sewers' services are consumed, and are of two kinds:

- "one-off": those connection and disconnection charges which cannot be recovered and transferred to other customers, e.g. the labour costs of laying a service pipe or installing a meter.

- "continuing" this category includes both regular impositions to do with the maintenance of the connection, reading meters, billing, collection of charges and various other consumer services; and also the costs of equipment (like meters) which can in principle be transferred to other customers should a particular customer disconnect from the system.

Commodity costs are also known as operating costs and are those costs that vary directly with the number of water units consumed, and certain other characteristics of those units, e.g. strength of sewage. They mainly include pumping costs and chemicals.

Capacity costs are those costs incurred in the provision of resources, distribution, administration, local storage, sewerage, treatment work, etc. Often they vary with one or other of the maximum demands made on the supply or sewage disposal system. Capacity costs generally account for more than 50 percent of resource-use costs.

A3.3.4 Natural Resource-Depletion Costs

Natural resource-depletion costs are those incurred in the maintaining of the quality and reliable availability of the basic water resource - river flow, ground water table etc. For example, if a municipality pumped water from a ground water table, and the level of this table fell and it needed to pump water from a river to replace the water losses, the natural resource-depletion costs would be the costs of diverting, storing and treating the river water.

A3.3.5 Damage Costs

Damage costs are those costs related to the environmental deterioration of the water service or of the surrounding site and which have adverse effects on the enjoyment or production possibilities of other water service users.

A3.3.6 The Calculation of Costs

A3.3.6.1 *Marginal Costs*

The marginal cost pricing system requires that prices should reflect the incremental costs imposed on the economy in satisfying marginal demands.

The marginal values of the costs presented above should thus be calculated.

A3.3.6.2 *Resource-Use Costs*

Customer Costs

a) *"one-off"*

The marginal cost of one-off charges is very close to the average cost because of the nature of the service. One could therefore approximate it by using the average cost. Payment should ideally be of a capital some of money. The inability of future customers to access capital may inhibit joining/leaving activities. A case can thus be made for annualising, or at least spreading over a number of years, "one-off" charges, effectively adding them to the fixed element of the period bill.

b) *"continuing"*

The marginal cost of "continuing charges" may vary by consumer group, (defined by the size of connection, the type of meters, the distance from the next point of metering, etc.). Within each consumer group the marginal cost of supply differs very little from the average cost, thus the average cost may be used instead of the marginal cost. However, a detailed classification of consumer groups requires administrative costs and therefore should only be undertaken if the benefits outweigh the administrative costs of its introduction.

Commodity Costs

Marginal commodity costs should be estimated as the anticipated increase in annual commodity costs divided by the anticipated increase in the annual quantity consumed.

Capacity Costs

> The essence of the methodology is to look ahead at the water authorities' or undertakings' actual or potential investment programmes to deal with expansion and deduce as much as possible about the extra costs that extra demands threaten to impose on the supply system. In OECD (1987, Annex 2) and OECD (1989b, Annex A) there is a full presentation of the ways in which the extra costs may be specified, measured and related to the extra demands. Both annexes are attached to this report under annex 1 and annex 2 respectively.

A3.3.6.3 Natural Resource-Depletion Costs

"Resource-depletion costs, both quantitative and qualitative, can in principle and in practice be estimated by valuing the extra resources required to maintain the quantitative and qualitative dimensions of the basic resource." (OECD, 1989b). Marginal resource-depletion costs are thus calculated by spreading the resource-depletion costs over the incremental output of the water service under consideration.

A3.3.6.4 Damage Costs

Damage costs are very difficult to estimate, because the environmental services such as amenities, maintenance of the ecological balance of the ecosystems, etc., which water provides, are either not marketable, or only partly marketable. Therefore, the deterioration of the environmental services water provides (the damage), is either unpriced or underpriced.

A number of methods, like the travel cost method, the hedonic pricing approach or, the contingent valuation method, have been developed for evaluating damage costs. However, although these methods succeed, more or less, in approximating damage costs, their informational requirements are such that it is practically impossible to find the true damage costs.

Damage costs related to environmental deterioration or destruction of a water service other than pollution, for instance the disappearance of a watercourse (because of a diversion) which has amenity value, should be taken into account in the decision-making process. When an investment decision is considered, damage costs, estimated by one of the evaluation methods mentioned above, should be taken into account in the cost-benefit analysis. If there is a well-established public water service sector already in place, damage costs should be taken into account by embodying an allowance for them in the pricing system of the water service sector under consideration.

Damage costs related to pollution should bear a charge per unit of pollution discharged.

If the marginal damage costs were known they could be reflected in the prices by putting a charge (a price) per unit of pollution discharged equal to the marginal damage cost.

It has been shown that when marginal damage costs are not fully estimated, but there are some indications regarding their possible level, then there is still room for promoting economic efficiency and environmental effectiveness. In fact, if a charge per unit of pollution discharged is levied at a high enough level, that polluters have an incentive to abate, then a reduction in the pollution levels may be expected. This pollution reduction will almost certainly not bring pollution to the optimum levels (i.e. the pollution levels corresponding to a situation of efficient allocation of resources) but it will be a move in the right direction.

In order to encourage economic agents to respond to environmental charges by reducing their pollution levels, one condition is essential: the increases in environmental charges should cause significant reductions in pollution levels, i.e. price elasticity should be significantly different from zero. If increases in environmental charges do not induce polluters to reduce their pollution levels, then environmental charges can not promote efficient allocation of resources. They can only be used for revenue-raising purposes.

The elasticity condition is required for the marginal cost pricing as well. If economic agents do not respond to tariffs based on the principle of marginal cost pricing it is not worthwhile allocating funds for devising, implementing, administering and updating such complex tariffs.

In the second part price elasticities will thus be given particular consideration.

A3.3.7 MARKET-ORIENTED TECHNIQUES. SECTORIAL ANALYSIS

This part deals with the implementation of the market-oriented techniques by the water sector.

A3.3.7.1 The Public Water Supply

The public water supply sector deals with the provision of potable piped water. The potable piped water is mainly consumed by households and, to a lesser extent, by industries.

It is generally recognised that the pricing system of the public water supply sector is not related to the marginal value of supplying this water and that this has led to the over-building of systems, waste of public funds and sub-optimal water use practices (OECD, 1989b).

OECD (1987) reports the results of fourteen urban public water supply studies, reproduced below in Table A1, covering Australia, Canada, England and Wales, Finland, the Netherlands, Sweden and the United States. The result is clear-cut: with one exception (that for industrial demands in Rotterdam) price elasticities for year-round or off-peak use are different from zero. They are in the - 0.1 to - 0.3 range.

Such elasticities are not very high but still they are significant.

Therefore, it is advisable to proceed to estimations of price elasticities before deciding whether or not a charging system based on marginal costs should be established. We need to know whether price elasticities are high enough to successfully undertake a charging system for the management of public water supply.

If price elasticities reveal significant values then the marginal cost pricing should proceed in the way described above.

For resource-use costs, customer ("one-off" and "continuing") commodity and capacity costs need to be estimated.

For the calculation of marginal capacity costs there might be room for taking into account peak demands as well.

Country	Location	Type of Study	Estimated Price Elasticity	Reference
Australia	971 households in 20 groups in Perth	readings over 1976-82; pooled x-section and time series	overall: -0.11	Metropolitan Water Authority, 1985
Australia	315 households in Perth	x-section (hypothetical valuation technique)	in-house: -0.04 ex-house: -0.31 overall: -0.18	Thomas, Syme and Gosselink, 1983
Australia	metered	x-section (?)	winter: -0.36	Gallagher and Robinson, 1977
Australia	137 households in Toowoomba, Queensland	1972-3 to 1976-7 pooled cross-section and time series	short-term: -0.26 long-term: -0.75	Gallagher, et al, 1981
Canada	urban demand, eastern Canada	x-section, 1960s	winter: -0.75 Summer: -1.07	Grima, 1972
Canada	municipal demand, Victoria, B. C.	time series, 1954-70	winter: -0.58 summer: zero mid-peak: -0.25 year-round: -0.40	Sewell and Roueche, 1974
England and Wales	all firms in Severn-Trent	water-saving investment in 1972-78	-0.30	Thackray and Archibald, 1981
England and Wales	industrial (metered) consumption England and Wales	time series 1962-80	year-round: -0.30	Herrington, 1982
Finland	municipal demand Helsinki	time series, 1970-78	year-round: -0.30	Laukkanen, 1981
Netherlands	industrial demand, Rotterdam	time series, 1960s and 1970s	"no price elasticity demonstrated"	Rotterdam Water Authority, 1976
Sweden	69 domestic residences in Malmo	14 readings each over 1971-78; pooled cross-section and time series	year-round: -0.15	Hanke & de Maré, 1982
United States	2159 households in Tucson, Arizona (water use per household)	42 readings each over 42 months, July 1976-Dec 1979; pooled cross-section and time series	year-round: -0.256	Martin, Ingram, Laney & Griffin, 1983
United States (1)	domestic use in Tucson, Arizona	time series, Jan 1974 - Sept 1977	year-round long model: -0.27 linear: -0.45/0.61	Billings and Agthe, 1980
United States (2)	residential use in 21 study areas, eastern and western United States	x-section, early 1960s	winter: -0.06 summer (east): -0.57 summer (west): -0.43	Howe, 1982

1 Price included volumetric price of sewer use and the whole tariff schedule (increasing block was assumed to change in the same proportion as "marginal rate" changes).

2 Changes in marginal price (= marginal block rate) only, although intramarginal rate structure allowed for in demand function. These elasticities represent significant reductions on these estimated from the same data fifteen years earlier (when the intramarginal rate structure was not allowed for): -0.23, -0.86, and -0.52, respectively (see Howe and Linaweaver, 1967, and Howe, 1982).

TABLE A1: PRICE ELASTICITIES FOR URBAN WATER SUPPLY (Source: OECD 1987)

On the one hand, according to OECD (1989b) price elasticities of water demand for ex-house use, e.g. watering gardens, are high. These demands arise mostly in summertime. Moreover, a number of US. studies, presented in Table A2, which refers to the 1970s, showed peak day ratios down 10 percent following the introduction of seasonal tariffs. Also, in Antwerp, Belgium, since industrial consumers have been able to take advantage of a complex night/day tariff, the peak hour ratio has fallen from 1.6 (1963) to 1.1 (1985) (OECD, 1987).

On the other hand, although technological progress allows for more accurate measures of time-related consumption, monitoring still remains expensive and difficult.

Therefore, when determining the marginal capacity costs, peak demands should only be taken into consideration if there is reason to believe that price elasticities are high enough. This will ensure that the benefits in efficiency and environmental effectiveness outweigh the costs of identifying and measuring the marginal peak demands.

Regarding natural resource-depletion costs, it should be noted that they might be significant. Increasingly, potable water is abstracted from ground water bodies and the costs for maintaining the quality and quantity of ground water are higher than the equivalent costs for surface water (ground water is limited and of higher quality).

As regards damage costs attempts should be made to evaluate them if thought to be significant. The various available methods for evaluating damage costs should be used. As already explained these methods can only indicate the size of damage costs. They cannot give a precise estimation of the damage costs. Thus, damage costs should be taken into account in the pricing system by simply embodying an allowance for them.

It is believed that if resource-use costs as well as natural resource-depletion costs have been properly taken into account damage costs would be minimised. If water becomes expensive then overuse of water will be significantly reduced and consequently damage costs will be minimised.

Water Authority	Year of Intro-duction	Details of Scheme	Outline Effects	Reference
Fairfax County, Virginia	1974	Peak use charge of $2.45 per 1000 US galls (3 785 l) on all 2-qtr. use greater than 1.3 times winter qtr. use. Ordinary commodity charge ≈ 70c. per 1000US galls.	1974-80: fall in consumption; peak day ratio fell 1.63 to 1.4 (similar climate) 1974-/1977.	Griffith (1982)
Dallas, Texas	1977	For each consumer, surcharge of $0..5/m^3 (= 31%) in all June to Sept consumption above 120m^3 per month.	1977 peak day ratio fell from 1.97 to 1.8 (climate 'more adverse'). Peak day demand fell by 12% on previous 5-year maximum.	Rice & Shaw (1978)
Tucson, Arizona	1977	High-tranche summer tariff 60% above low-tranche summer and constant winter tariffs (residential). Differential 40% to 70% summer mark-ups for other consumer groups.	1980 peak day demand down 25% on 1976. Average daily demands down by 15% (but simultaneous 'Beat the Peak' campaign in operation).	Zamora, Kneese & Erikson (1981)
Santa Fe, New Mexico	1978	Summer / winter differ-entials of 5% to 70% super-imposed on existing decreasing-block rates.	1975-79: average daily per capita demands declined 16%.	Zamora, Kneese & Erikson (1981)

TABLE A2: SEASONAL TARIFFS IN THE UNITED STATES

Source: OECD (1987).

Examples of marginal cost pricing in public water systems are not available. Those who are interested in examples of other charging systems they can find them in OECD (1987).

A3.3.7.2 Direct Abstractions

This section deals with the direct abstractions (or withdrawals) of water from its sources.

Direct abstraction is made by municipalities, industry and agriculture. Municipalities require direct abstraction in order to fulfil their public supply function. The costs related to direct abstractions are taken into account through the public supply system. Thus, municipalities will not be considered in this section.

A3.3.7.2.1 Industry

There are no empirical estimates of price elasticities in this area. Evidence concerning the incentive effects of direct abstraction charges is also very limited and relates only to industrial water consumption in France (OECD, 1989b) and to industrial location decisions in Japan and France (OECD, 1980).

As far as industrial water consumption is concerned, the Picardy Agence de Bassin imposed, in 1970, charges on the direct abstractions of water, after having found that the water table at Lille was falling significantly. This decision, together with other factors, led to industrial water consumption falling by 50 per cent ten years later.

Regarding industrial location decisions it was tentatively suggested in OECD (1980) that in the Seine and Yodo River basins "the incentive effect (of abstraction charges) may be considerable. They may induce thermal power stations and industries which are big users of water to avoid certain zones where abstraction rates are high."

Although there are no empirical estimates of price elasticities the above examples, taken together with economic theory, let us conclude that price elasticities can take on significantly negative values. "For surely, just as there are low prices of directly abstracted water at which an industrialist does not find it worthwhile to install recirculation equipment, there are higher and higher prices at which it becomes worthwhile to install increasing amounts of such equipment. This results in higher and higher water re-use coefficients." (OECD, 1987).

Marginal cost pricing should therefore be promoted and should be calculated as follows.

Resource-use costs: the only resource-use costs involved are metering and extra billing.

Natural resource-depletion costs: they can be very significant for the case of ground water abstractions and of lesser importance for the case of surface water abstractions.

Damage costs: same remark as above, although for big diversion schemes damage costs related to surface water abstractions can be very significant.

Two remarks pertinent to direct abstraction charges should be made:

The first concerns the manner in which direct abstraction charges should be levied. Ideally, charges should accompany a well-established system of authorised and actual abstractions monitoring. Authorised abstractions should reflect the marginal natural resource-depletion costs and marginal damage costs, since total authorisations determine the natural resource-depletion and damage costs. Charges for actual abstractions should reflect the marginal operating costs of the system, mainly costs to the water authority of meter reading and extra billing. (OECD, 1987 and OECD, 1989b).

The second relates to summer/winter, or other seasonal, distinctions in charging for direct abstractions. For many surface waters, capital works are necessary to support only summer flows. For these cases the marginal capital costs of the support works should be presented, via prices, to the authorised abstractions of the summer users. For ground water, no summer/winter distinction in charging should be made. Percolation patterns are long and often unknown. Thus, ground water abstraction should be considered as non-seasonal.

A3.3.7.2.2 Agriculture

Irrigated agriculture is one of the largest users of water in South Africa and one of the most highly subsidised. Subsidisation affects mainly two levels of agricultural production: agricultural prices and water prices (mainly via subsidisation of irrigation schemes). It induces overproduction of agricultural products (which in turn implies overuse of water) and excessive use of water. This excessive use of water may in turn result in increased nitrate, phosphate and pesticide contamination of acquifers.

The first step in order to promote economic efficiency and environmental effectiveness should thus be to remove all these subsidies. If, at present, it is politically unacceptable to reduce subsidies on agricultural prices one should start by gradually removing the subsidies on water prices (mainly on irrigation schemes).

Price elasticity estimates for irrigation demand show high values. Cross-sectional studies in the United States and Australia, presented in the table of the next page, reveal "below-average-price" elasticities of demand for various crops in the 0.1 to-0.5 range and "above-average-price" elasticities in the 0.5 to 2.3 range.

Thus the second step in order to promote economic efficiency and environmental effectiveness is to proceed to marginal cost pricing.

Resource-use costs: they involve customer (one-off and continuing) and capacity costs.

Natural resource-depletion costs: they can be very significant for the case of ground water abstractions and of lesser importance for the case of surface water abstractions.

Author	'Average' Elasticity	'Low-price' Elasticity	'High-price' Elasticity	Area studied
I. CALIFORNIA, USA, 1960s AND 1970s				
Moore	-0.65	-0.14	-1.58	San Joaquin, Calif. (linear regression)
Moore/Hedges	-0.65	-0.19	-0.70	Same (quadratic regression)
Bain/Caves/Margolis	-0.64	-	-	34 Calif. water districts
Heady et al.	-0.37	-0.17	-0.56	17 US western states
Shumway et al.	-	-0.56	-2.32	Calif. (2-eqn. model)
Shumway et al.	-	-0.48	-2.03	Calif. (1-eqn. model)
Howitt/Watson/Adam	-0.97	-	-	California (linear programming approach)
Howitt/Watson/Adam	-1.50	-	-	California (quadratic programming approach)
II. AUSTRALIA, 1964				
Flinn				
Total seasonal demand	-0.46	-0.09/-0.25	-0.91/-1.73	5 representative farms in Yanco Irrigation Area (linear programming approach)
Spring only	-0.70	-0.09/-0.26	-1.61	
Summer only	0.06	-0.01/-0.03	-0.09	
Autumn only	-0.68	-0.09/-0.25	-1.56	

TABLE A3: CROSS-SECTIONAL PRICE ELASTICITY ESTIMATES FOR IRRIGATION DEMANDS

[Source: Anderson (1983), chapter 3, for US data;
Flinn (1969), for Australian data.]

Damage costs: same remark as above, although for big diversion schemes damage costs related to surface water abstractions can be very significant.

The two remarks made in the industry sub-section (A3.3.7.2.1) apply here as well.

One example of marginal cost pricing for the abstraction of water services has been found. Although it neither reveals the way the marginal cost pricing has proceeded nor the results, we present it as the only available example in this area.

In 1970, the Société du Canal de Provence et d'Aménagement de la Region Provencale, which supplies 60,000 hectares of farmland and nearly 120 communes, initiated a charging scheme based on the theory of marginal cost pricing (Jean, 1980). This scheme provided full recognition of the need to consider and reflect long-run costs if farmers are to make "correct" investment decisions in terms of land, cultivation, crops, irrigation equipment and storage. A peak period is identified lasting for four months from mid-May to mid-September which is a central component in the tariff structure. Tariff design starts from the objective that tariffs should reflect:

- In the peak period, long-run marginal capital costs augmented by operating costs, and
- In the off-peak period, operating costs only. (OECD, 1987).

A3.4 WATER PRICING - "TRANSITION" [1]

Economic efficiency will be achieved when the necessary assumptions underlying the formation of a competitive market are met, and such a market will fulfil the optimal conditions for both consumers and suppliers of water. To achieve this, the rate of water substitution in production must equal the rate of water substitution in consumption and each is equal to a common price. When this equilibrium position is reached a further reallocation of water would make some consumers worse off than they are at the equilibrium price and such a reallocation is therefore undesirable. The important results arising from this are then:

1. Competitive equilibrium satisfies the commonly held definition of economic efficiency, and
2. An equality between the marginal costs of supplying water and corresponding prices for water fulfils the Pareto optimality conditions for economic efficiency.

The absence of a market for water in South Africa precludes these conditions coming about. This problem can be overcome by simulating a market to price and allocate water. To ensure that conditions 1 and 2 above are met in this simulated market, thus ensuring economic efficiency in the pricing/allocation process, economic theory demands that, because there is no free competition between many

1 This section is based upon a model developed by GA Veck and DJ Stevens.

buyers and sellers of water, the price which the monopoly supplier (in South Africa this is the DWA&F) charges for water must be the marginal cost of supply. This price in turn must equal the value of the last unit of water purchased and used by each of the different water consumers of the South African water economy thus, the second condition stipulated above for economic efficiency is met. So far as condition 1 is concerned Samuelson (1952)[2] showed that competitive equilibrium in a simulated market would come about between prices and commodity allocations between economic sectors when, what he termed Net Social Payoff (NSP), was maximised. In the model developed here the commodity allocated between different sectors is of course water. Now NSP is the sum of the social payoffs throughout the economy minus the transport costs involved in moving the commodity between different sectors. Social payoff for any sector is defined by Samuelson as "the algebraic area under the *excess* demand schedule", that is, the area under the demand schedule less the area under the supply schedule. This excess area can be considered to be a measure of the social welfare referred to above. The next section discusses this in detail.

A3.4.1 Social Welfare and its Measurement

In normative economics the value of a commodity is determined by the amount of money a person is willing to pay for it. Now in the past in South Africa the price paid for water has been a rather unstable measure of its real economic value. That this has been a common trend over time and in different places was ably demonstrated by Adam Smith when he contrasted the price of water, which was cheap relative to its life supporting qualities, with the price of diamonds which had no biological value at all, but were highly priced[3]. This paradox is easily explained and results from the fact that demand is a function of marginal utility and not total utility.

In considering the worth of a commodity, if a person is prepared to pay, for example, R3 for a litre of water that is the value he attaches to water. If he has to pay R1 for a litre of water, however, this price provides him with a surplus of R2 to spend on other things. The surplus is the amount of money a consumer would be prepared to pay for a given amount of a good, less that which he actually pays. This is directly related to Samuelson's Net Social Payoff as defined above (Samuelson, 1952,

2 Samuelson, Paul, A., Spatial Price Equilibrium and Linear Programming, American Economic Review, 42, 1952, pp. 283-303.

3 Smith, Adam.: The Wealth of Nations, Book 1, Chapter IV, London, 1776.

p. 28). This idea of a surplus may be extended by determining the amount of money a consumer would be willing to pay for one litre of water, then for another and so on. These sums of money can be regarded as marginal valuations and can be plotted in histogram form denoting a demand schedule[4]. If the price of water is fixed at R0,50/litre a consumer will continue to purchase water until its marginal value is equal to, or below, this price. If in Figure A1, P_1 represents this price then the shaded area above P_1 is equal to the net social payoff as defined by Samuelson. It must be remembered that if economic efficiency is to be achieved in the simulated market for water, the horizontal line (price P_1) is the marginal cost for supplying the water.

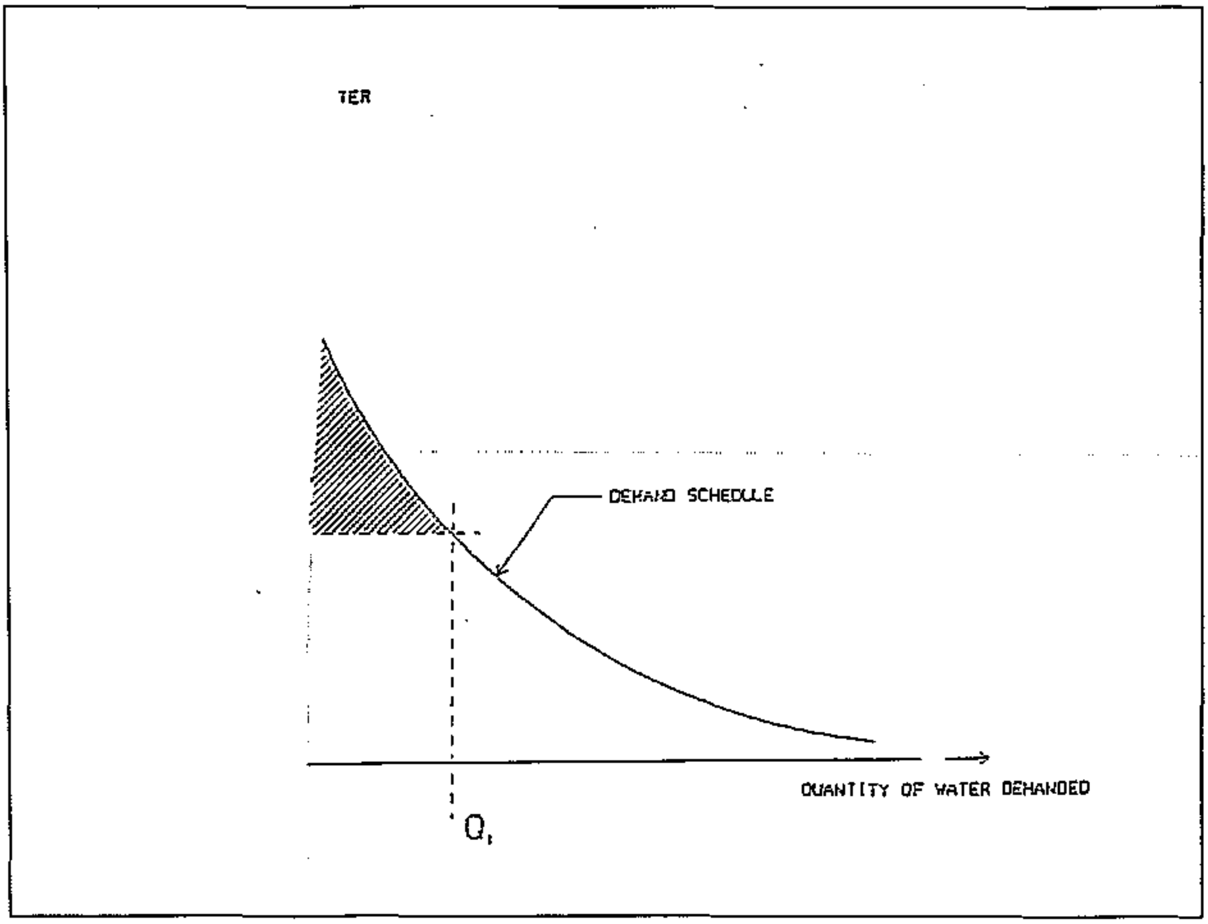

FIGURE A1: DEMAND SCHEDULE SHOWING AREA OF NET SOCIAL PAYOFF

4 Once perfect, divisibility is assumed the stepped outline of the demand curve becomes a smooth curve as depicted in Figure A1.

Conventional interpretation of the demand schedule is amended so far as measuring social payoff is concerned. Instead of reading the demand schedule horizontally it is read in the vertical direction, i.e., given an amount of water on offer, the corresponding price is a measure of the consumer's willingness-to-pay. As apposed to starting with a particular price and the demand schedule being a measure of the water demanded. The market demand curve is clearly the horizontal summation of all individual demand curves, e.g., all irrigators, or domestic water users, etc., and can be regarded as the marginal valuation curve of water for society.

To illustrate the importance of the concept of social payoff to society any investment in a water scheme which has the objective of reducing the cost of a good or service confers a benefit on society as a whole. This can be demonstrated with reference to Figure A2. In this example let the benefit to a community from extending an existing irrigation scheme be the increase in agricultural output arising from more water being available to expand the crop growing area. Prior to building the extension to the irrigation scheme the price of agricultural produce is represented by area ABP2. If the irrigation scheme extension enables farmers to reduce the price of their agricultural produce from P_2 to P_1 (due to increases production and economies of scale) the consumers' social payoff increases to A C P_1 i.e., an increase equal to the shaded strip P_2 B C P_1. This increase in social payoff is made up of two components. First, there is the cost saving component, the rectangle P_2 B D P_1, which represents the savings per unit of agricultural produce bought multiplied by the original amount of produce bought, Q_1. Secondly, there is the increased amount of produce bought as a direct result of the extension of the irrigation scheme which in turn allows the farmers to reduce the price of their produce. This second component is represented by the triangle BCD. It is the social payoff arising from the extra amount of agricultural produce bought, i.e., Q1Q2 either by the original consumers or by new consumers who enter the market as a consequence of the fall in the price of agricultural produce from the extension to the irrigation scheme.

It is important to note that this social payoff is a lower bound of benefits since no account has been taken in the explanation given above of the additional benefits accruing to the economy from the interrelationships which exist between different sectors of the economy. In other words, because of increased economic activity resulting in the enlargement of the irrigation capacity of their farms, farmers are now able to purchase more equipment to handle the increased agricultural output. They may also employ more people to gather crops. Additionally, where crops are processed and canned larger facilities may be built to deal with the increased throughput.

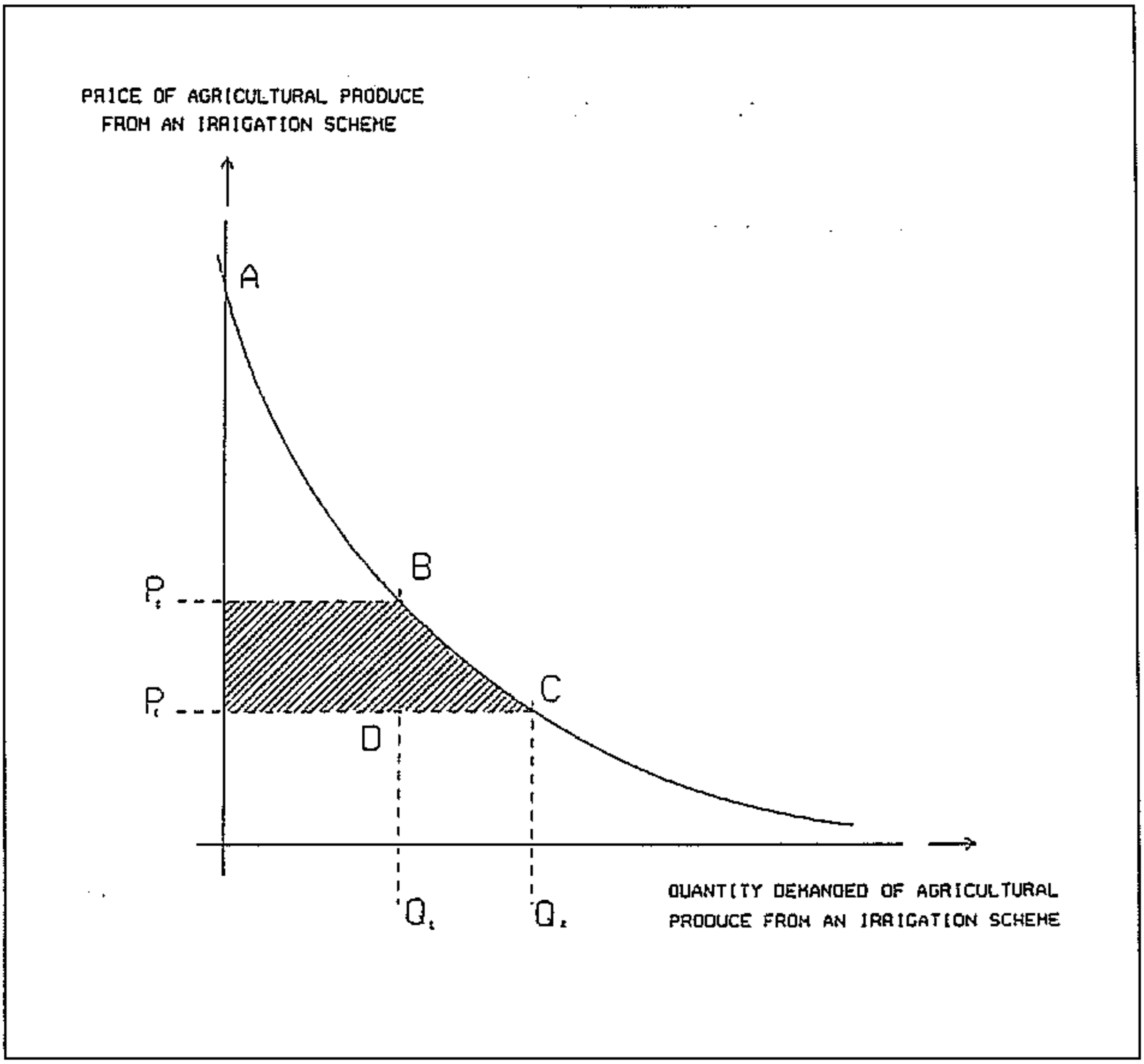

FIGURE A2: ADDITIONAL NET SOCIAL PAYOFF FROM EXTENDING AN IRRIGATION SCHEME

Now Samuelson's social payoff is derived from what Marshall (1947) called consumer's surplus, i.e., the areas under the demand schedule for a particular good as defined by the shaded area of Figure A1. This area is a sufficiently close approximation of the net size of the total social benefits or social welfare derived by any sector of the economy under a particular pricing regime (cf., Bohm, 1987, p. 50, Willig,1976, p.589, Margalin, 1962, pp., 36-52)[5]. Harold Hotelling also wrote that consumer's surpluses "give a meaningful measure of social value". The purpose of the model developed here will be to maximise this social welfare in an economically efficient manner in the South African water economy. It is to be noted that the explanation of social payoff given here does not mention utility since the approach taken transfers utility into monetary terms.

To meaningfully maximise social welfare in the South African water economy, it will require that certain physical characteristics of water be allowed for in the construct of the model. These characteristics are discussed later below.

The model developed in these sections has sought to provide water planners and administrators with the means of pricing and allocating water in an economically efficient manner. It also takes into account the four principle physical characteristics of water which have been shown to govern the water management process. These characteristics are: temporal, quantity, location and quality. The model developed focuses on the demand-side pricing and allocation of South Africa's water economy and attempts to maximise the economic benefits, or social welfare, that water can provide to the community as a whole.

A3.4.2 Physical Characteristics Of Water Which Impact Water Modelling

This section considers characteristics which impact the water pricing and allocation process so far as modelling the South African water economy is concerned. The first considered is time and how it can influence important parameters in the construct of a water pricing and allocation model.

5 Measuring the social payoff by the area under the demand schedule above, the prevailing price of water is an approximation, since the social payoff so found assumes that a consumer's indifference curves are parallel, which of course they may not be. The assumption is, however, sufficiently accurate to be used as a practical tool in economic analysis. In particular cf., Willig, RD.,. Consumer Surplus Without Apology, American Economic Review, 1976, pp. 589-597. This paper shows "that observed consumer's surplus can be rigorously utilised to estimate the unobservable compensating and equivalent variations - the correct theoretical measures of the welfare impact of changes in prices and income on an individual". The paper is drawn from doctoral research done at Stanford University by Willig.

A3.4.2.1 Temporal Characteristics

A distinction of importance in water modelling concerns the issues surrounding whether the price of water should vary between the long and short-term. The important point that has to be borne in mind in considering this problem, is that in the short-run the amount of water available can be considered certain and fixed. The amount of water available in the long-term is uncertain in South Africa. A rational consumer's willingness-to-pay for water is related to the price he obtains for his output from a production process in which water is a factor of production. Consequence, in the short-term, where water quantities are fixed the increase in net value of output can correctly ignore the sunk costs associated with the water investment. In the long run, however, such costs must be recovered. The price a consumer will pay for water will, as a result be affected.

When attempting to determine prices for water and its allocation over time to competing end users establishing some common time span is desirable. Here data availability becomes important and can dictate the period chosen. The period chosen should not, however, be so long that it conceals, rather than highlights the competitive and complementary relationships which exist between end users of water.

A3.4.2.2 Instream and Off-stream uses

The economic characteristics of water must not only be compared in time, if prices are to have any meaning and be strictly comparable, but also in use. The optimal price which can be charged for water is dependent therefore on its physical usage for economic purposes. In this respect it is common to distinguish between in-stream and off-stream uses. In-stream uses are generally non-withdrawal uses whilst off-stream uses are concerned with the withdrawal of water from its source for use in a production process. Uses of water for the purpose of recreation, hydro-electric generation, fishing, wildlife support and waste assimilation are categorised as in-stream uses as a consequence. Water used in various production processes for industry and agriculture (e.g., irrigation use) are categorised as off-stream uses. A pricing/allocation model for water should ideally have the facility to include the in-stream and off-stream distinctions in its construct.

A3.4.2.3 Stochastic Considerations

In the pricing/allocation model developed here, a major assumption is that all system inputs and future demand parameters are deterministic. Although deterministic analysis gives perfectly useful results for water resource planners and managers in truth, water system inputs are really stochastic in nature. In South Africa frequent droughts often render water supplies different from those expected.

An important consequence of water supply reductions is that production processes are organised in a less efficient manner than would have been the case where accurate forecasts of supply could have been made. Deficits during a crop growing season for example would be more costly than if such reductions had been planned in advance. In a similar way economic problems occur when supplies exceed those forecast: here the resulting benefits would be less than those which would have been gained if the increased supply of water had been known and planned for in advance.

Because of the stochastic nature of water supply in South Africa, water models should ideally be constructed to allow risk and uncertainty to be factored into analysis by parametric changes in the input data of such models. Logically, it seems that three strategies can readily be employed to counter stochastic stream flow and demand etc., these are: **(i)** simulation analysis where historical parametric sequences are used and which assume that what occurred in the past will occur in the future; **(ii)** simulation analysis using a number of postulated scenarios of possible future occurrences. Here data can be obtained by using probability analysis for determining distributions of input parameters to the economic model so that equally likely future scenarios are catered for in the water planning process, and **(iii)** a restricted simulation to limit analytical costs and extensive and expensive data gathering. The obvious problem with approach (iii) is that after a future possible set of stochastic data has been formulated the event simulated may or may not occur and other events, covered in simulation (ii) may eventuate resulting in such things as flood damage etc., which, as a result of a water resource scheme being built around approach (iii), the chosen planning regime was quite unable to cope with.

A3.4.2.4 Location issues

Water supply costs which include the costs of storage, treatment and transportation are intimately related to the physical location of bodies of water and watersheds from which supplies to irrigation, industry and municipalities are drawn. Watersheds or drainage basins are land units which gather runoff and collect precipitation which then usually form streams which often merge as rivers and eventually discharge into the sea. In the process, however, they form the bodies of water mentioned above.

Water from watersheds has to usually be diverted to areas where man can use this water. Therefore, pricing/allocation models must, along with the other issues mentioned already, be constructed so that the costs involved in such diversions or transportation are captured in the pricing process.

A3.4.2.5 Water Quality Issues

The quality of water required in different uses may vary considerably, and can also affect the quantity demanded. The lowest common denominator for specifying quality is the water quality found in the water body from which withdrawal for different uses takes place. Water quality considerations are therefore an integral part of water resources planning. The problems of water quality are irrevocably tied to that of water quantity availability for certain uses, e.g., potable water. In water quality control there may be important objectives in addition to, or in lieu of, pure economic efficiency, e.g., political or social factors could in certain instances become dominant. In the model developed in this section only measurable parameters are considered that can be quantified in terms of biochemical oxygen demand (BOD)[6]. In modelling water pollution effects it is important to evaluate the impact water pollution has on water price since without such an evaluation it is difficult for water resource planners to justify a particular water scheme in terms of national economic benefits, e.g., the impact water pollution has on industrial or agricultural output.

There are numerous methods for controlling water quality levels, but incorporating methods of treatment into an economic model for pricing and allocating water adds complexity into an already complex problem. As a first assumption, the cost of water treatment can usually be expressed as a function of the quality of raw water. Considerable data is required to incorporate meaningful quality constraints into an economic model for pricing and allocating water.

Events such as those described above create difficulties for water managers and theoretical problems for economists developing pricing/allocation models for water resources. Consequently, and in an attempt to take these physical characteristics into account in an economic model, for water resource management the entire water

6 BOD is the total oxygen requirement for the oxidation of biodegradable organic material contained in a waste stream. The constituents of the BOD consume dissolved oxygen (DO) in natural water courses. The DO is therefore one of the principal indicators of water quality. BOD-DO relationships can be said to govern then the management of water quality.

catchment or river system has to be considered. It is imperative therefore that water resource modelling should be approached in a systems context.

Approaching the economic analysis of pricing and allocating water from the systems context involves nothing more than simulating the catchment area or water course by mathematical formulation subject to optimisation techniques almost invariably with the use of computers. It is essentially a combination of response and accounting methodologies. The response portion takes into account the physical economic and social elements making up the economic analysis whilst the accounting portion puts monetary values to the elements. The final result is one of detailing the consequences of the particular water investment in terms of economic costs and returns on the investment.

The end goal of the systems approach to demand-side water resource investments is that a faithful reproduction of the investment is made which realistically portrays the real world and the consequences in economic terms of certain actions taken by water resource planners and managers. The great contribution to be gained from the systems approach is one of evaluation and also the creation of ideas for modifying a particular water investment at its design stage so that the greatest benefits are obtained.

The systems approach allows the analyst to examine many alternatives and helps administrators formulate policy. In this respect systems analysis is important in water planning and the operation of a particular water scheme so that maximum benefit may be obtained for the community as a whole. So far as planning is concerned the development of a water catchment can be carried out in a composite manner in which the location of dams can be decided upon; where new irrigation areas should be planted and even where new towns or smaller urban settlements should be built. All combinations of these possible alternatives can be considered by manipulating the construct of a pragmatic systems model for water resource investments.

In the simulated market for water, which is required for modelling purposes, the DWA&F and water boards supplying water to various consumers in South Africa are assumed to price and allocate water in such a way that they are seen to be concerned with the maximisation of social welfare to end users, rather than the maximisation of profits. Also, as mentioned above, competitive water prices and allocation can still be achieved, however, if the price of water to each class of consumer equals the marginal cost of servicing that particular customer. The requirement is of course that the DWA&F and the various water boards of South

Africa refrain form pricing and allocating water on the basis of non-market considerations.

To mirror reality in the simulated market for water, it is assumed in the model that water storage and distribution components of the water delivery system such as reservoirs, dams, pipelines, canals pumping stations, and water treatment plants are fixed in capacity. This means that the designed water capacity of the systems may limit the delivery of water to competing end-users in periods of peak demand. Economically speaking this will have the effect of forcing the marginal product of water above the figure which would prevail if the capacity of the water supply system had been larger. This is simply because in such a situation demand for water outstrips supply. Consequently, it is possible that the price for water will vary, not only between different economic sectors, e.g., irrigated agriculture, industry and municipalities, but also within these sectors. Such a situation would occur in cases where the curtailment of water supply at certain times forces the price for water above the prevailing price at non-peak demand times.

Demand schedules for water in different uses (they represent the willingness to pay by consumers for varying quantities of water), should be modified to take into account the treatment costs to purify water. The demand schedules in the model are therefore hybrid. This is demonstrated when water quality problems are considered in Appendix B. In the model developed here it is assumed that the demand schedules for water in its different uses are continuous (they need not be linear), differentiable and intercept both the price and quantity axes at definite values so that they are easily incorporated into the objective function of the model.

A3.4.3 The Mathematics of the Model

The mathematical formulation of the water pricing and allocation model will now be considered. The explanation is divided into two parts: the first part considers only the temporal, quantity and location constraints as they effect the model, the quality issue will be considered in the next section.

A3.4.3.1 Modelling the Temporal, Quantity and Location Characteristics of Water

First consider Figure A3. in which supply and demand schedules for water are contrasted. The supply schedule is shown as a horizontal line TZ where T is the marginal cost of water and depends solely on the quantity of water handled by the supplier i.e., the DWA&F or Water Board in question, and is assumed constant over the quantity of water delivered to a particular consumer. This marginal cost may vary from consumer to consumer and is considered to be made up of the components listed after Figure A3.

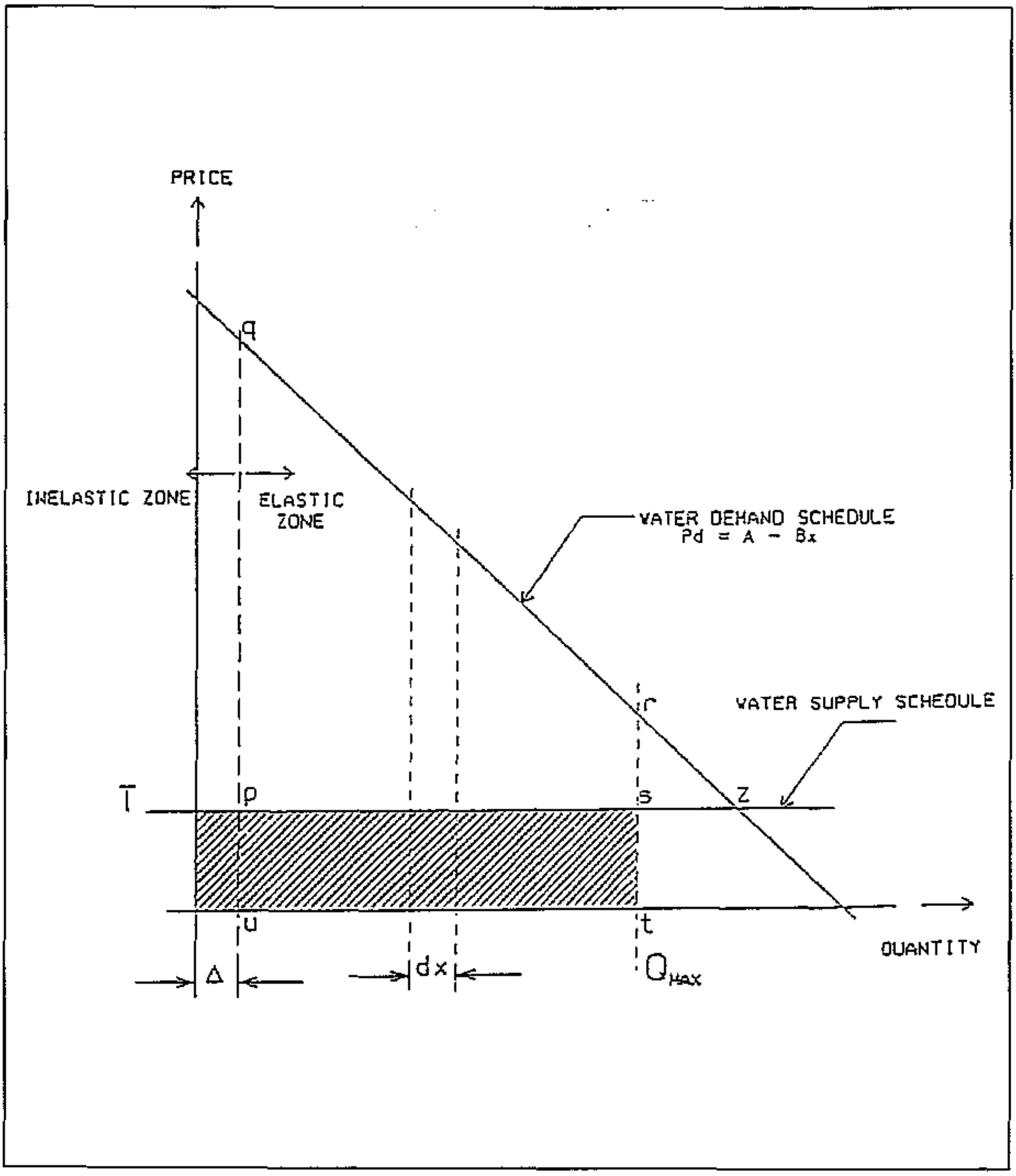

FIGURE A3: SUPPLY AND DEMAND SCHEDULES FOR WATER CONTRASTED

1. The costs of transporting water various distances to different consumers, i.e., the pumping costs per unit of water delivered.

2. The costs of building delivery works such as pumping stations, storage areas i.e., water towers in municipal areas etc., and filtration plants.

3. The costs of maintaining and replacing as the need arises, these water delivery works and storage areas.

4. The wages and salaries of staff and general administrative costs such as rentals, vehicle costs, etc.

5. Interest charges on the capital used to provide the water delivery system.

Costs 2-5 inclusive are considered to be shared on a pro-rata basis by each user whilst the transport costs vary from consumer to consumer. This means then different consumers pay different amounts for the supply of water as mentioned above.

So far as water transportation is concerned the smaller conveyance systems are usually pipelines requiring pumping plant although they can sometimes rely on gravity feed. Larger systems may have open canals or enclosed tunnels (the Fish River Tunnel in the Eastern Cape for example). Large delivery systems are usually associated with multi-purpose uses, for example hydro-electric generation, irrigation uses and flood control, for example the Drakensberg Pumped Storage scheme in Natal. Such delivery systems can include very expensive complex engineering works which have to be amortised over time, e.g., storage reservoirs to equalise water flow. Water supply systems are also influenced by the terrain over which they travel and consideration has of course to be given to water evaporation when the supply scheme has open channels. This problem is particularly important in South Africa. From the above it is perhaps easy to see that generalised water supply cost information is rather meaningless. This is why it was mentioned that delivery costs can theoretically vary from water consumer to water consumer and will be considered in further detail when a particular water catchment is analysed in the next chapter.

The water demand schedule is represented by the curve qrz shown here to be linear of the form Pd = A - Bx. Water demand is regarded in the model as occupying either an elastic or inelastic zone. The inelastic zone is the quantity of water demanded for life-support needs and conservation purposes. Other water demands may of course be added to this zone should the problem under review warrant it. The elastic zone is the quantity of water remaining after the inelastic water demand requirements have been deducted from the total quantity of water available for all purposes. The price axis in Figure A3. is adjusted by Δ to allow for inelastic requirements.

The quadrilateral pqrs is the area referred to earlier when Samuelson's social payoff was discussed and represents the total social welfare available to the consumer here considered. The market equilibrium price T shown in Figure A3 may, however, not be reached in practice due to a limit on total water availability. Q max, or limiting conditions in the water supply system or if water stored in dams and reservoirs is retained for various reasons such as a reserve for drought etc. It will be recalled that generally the demand for water is seasonal, particularly water used for irrigation purposes. The implication of this is that demand schedules can alter both in slope and intercepts depending on the relative scarcity of water and the particular season in which water is being priced and allocated to different consumers, i.e., summer, winter, etc.

The area representing social welfare, i.e., quadrilateral pqrs can be considered as a constrained optimisation problem as follows:

$$TSW(Q) = \int_0^Q (A - Bx)dx - TQ$$

$$= AQ - \frac{BQ^2}{2} - TQ \qquad \ldots\ldots\ldots (1)$$

Where Q is the quantity of water demanded and TSW is the total social welfare being equivalent to Samuelson's social payoff for a particular economic sector.

An additional constraint applies which is that $Q \leq Q$ max, i.e., the quantity of water demanded by the consumer must be less than the total water available in the system in any period, and of course the ability of the delivery system to supply that water.

Now generally the optimisation problem posed above will have more than one consumer and hence many variables will have to be dealt with in the optimisation process. Problems of this nature can conveniently be solved using the method of Lagrangian multipliers. This technique will therefore be used to solve the present problem.

Forming the Lagrangian function the problem may be specified as:

$$L(Q,P) = AQ - \frac{BQ^2}{2} - TQ + P(Q_{max} - Q) \qquad \ldots\ldots\ldots (2)$$

Differentiating L(Q,P) partially with respect to Q and P gives:

$$\frac{\partial L}{\partial Q} = A - T - BQ - P \qquad \text{.......(3)}$$

$$\frac{\partial L}{\partial P} = Q_{max} - Q \qquad \text{.......(4)}$$

The Kuhn-Tucker (1951) conditions[7], which strictly generalise the Lagrangian result to take into account inequality constraints are introduced for a solution. These conditions may be stated as follows:

$$\left(\frac{\partial L}{\partial Q}\right)Q = O \quad \text{and} \quad \left(\frac{\partial L}{\partial Q}\right) = O \quad \text{if} \quad Q \neq O \qquad \text{.......(5)}$$

$$\left(\frac{\partial L}{\partial P}\right)P = O \quad \text{and} \quad \left(\frac{\partial L}{\partial P}\right) = O \quad \text{if} \quad P \neq O \qquad \text{.......(5)}$$

From equations (4) and (5) $\quad Q = Q_{max}$ ....(6)

and $\quad P = A - BQ_{max} - T$ ....(7)

Now P is the Lagrangian multiplier, and is a price, since A and T which also multiply Q are prices. From equation (6) water, which has a price greater than T, must be used to capacity. The price P which can be interpreted as a water tariff amounts to a net excess over water supply costs. The analysis is now continued with reference being made to Figure A4. In this figure, quadrilateral pqrs is composed of two areas, A_1 and A_2. A_1 represents the benefits obtained from the consumer's social welfare whilst the area A_2 represent the excess benefits derived by the monopoly supplier i.e., the DWA&F or relevant Water Board. The benefits represented by A_2 will be re-invested in the economy thus increasing total social welfare. This comes about because, as will be recalled, the monopoly water supplier is deemed to be a non-profit making entity. T is of course an unavoidable cost of supplying water to competing end-users in the first place.

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7 cf., Kuhn, MW. and Tucker, AW. Non-linear Programming, Proceedings: Second Berkeley Symposium, J. Neyman, Ed., University of Califomia Press, 1951, pp. 481-492.

For a thorough discussion of the Kuhn-Tucker conditions, cf., Hadley, G., Non-linear Programming, Addison-Wesley, Reading Massachusetts, 1964.

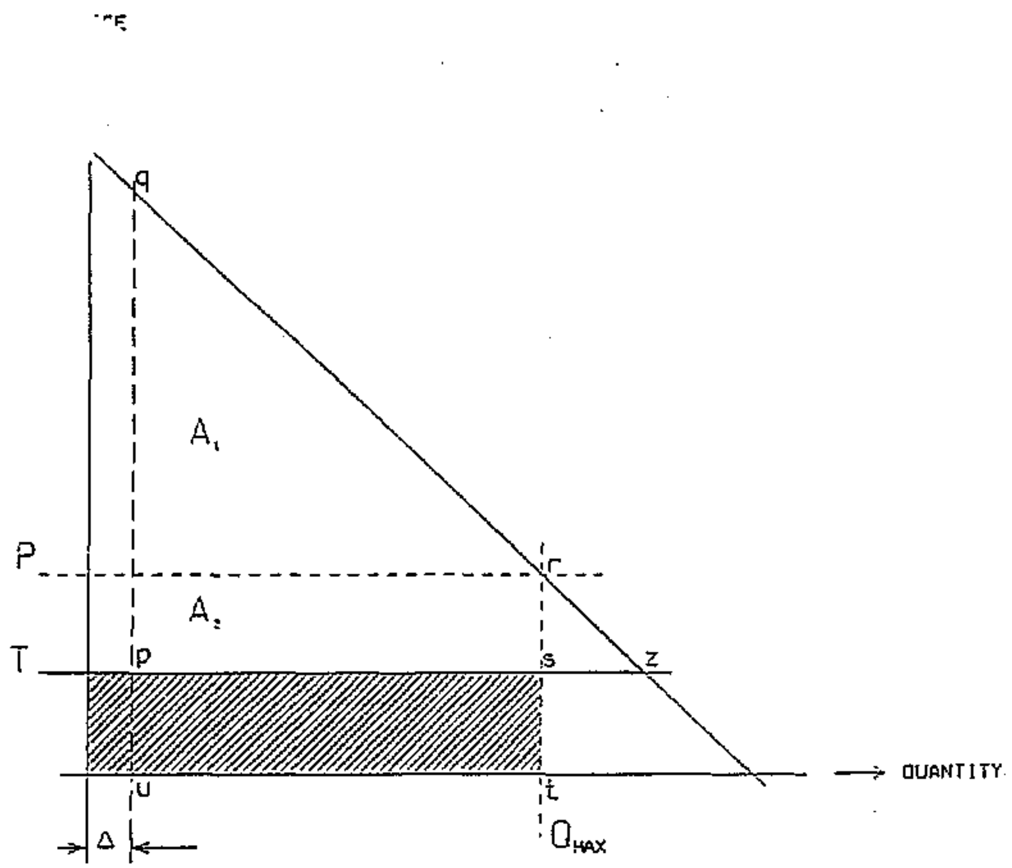

FIGURE A4: REPRESENTS BENEFITS OBTAINED FROM THE CONSUMER'S SOCIAL WELFARE THE EXCESS BENEFITS DERIVED BY THE MONOPOLY SUPPLIER OF WATER

The problem outlined above is the simplest possible case having just one consumer or sector of the economy demanding water. The problem is now generalised to one where several consumers compete for water. Here the price charged for water to each consumer will vary. To clarify this statement, the water demand price schedule $A - B_x$ will change from one water user to another. The amount Qmax will similarly vary since it will be limited by the capacity of the water delivery system to each user and also by the amount of water available in the whole system, e.g., irrigation conduits and channels and the volume of water stored in a reservoir supplying the systems with water. If the problem is considered over time, Qmax could also vary seasonally for a given user, at other times by drought conditions or an abundance of water in particularly wet seasons. The problem of maximising social welfare in an entire catchment area or river systems can be solved by individually adjusting the price and allocation of water to the various users.

The formal statement of the general problem in such a case and for ease of exposition limiting it to two water users or economic sectors is to maximise:

$$TSW = (A_1 - T)\,Q_1 - B_1\left(\frac{Q_1^2}{2}\right) + (A_2 - T)\,Q_2 - B_2\left(\frac{Q_2^2}{2}\right) \qquad \ldots\ldots (8)$$

subject to the following three constraints

$$Q_1 \leq Q_{1max}$$

$$Q_2 \leq Q_{2max}$$

$$Q_1 + Q_2 \leq Q_{Total}$$

where the symbols are as defined previously.

The constraints simply say that the supply of water to each of the two users cannot exceed the total amount of water available in the system. Furthermore, the supply to each user cannot exceed the amount of water that the delivery system can supply each user. The conditions apply in any time period since Q_1, Q_2 and Q total are seasonal quantities.

The Lagrangian function is again formulated for the complete catchment or river system and may be specified as:

$$\begin{aligned} L(Q_1, Q_2, P_1, P_2, P_3) = \;& (A_1 - T)Q_1 - B_1\left(\frac{Q_1^2}{2}\right) \\ +\;& (A_2 - T)Q_2 - B_2\left(\frac{Q_2^2}{2}\right) \\ +\;& P_1(Q_{1max} - Q_1) \\ +\;& P_2(Q_{2max} - Q_2) \\ +\;& P_3(Q_{total} - Q_1 - Q_2) \qquad \ldots\ldots (9) \end{aligned}$$

The Kuhn-Tucker conditions for a solution are formulated now as follows:

$$\left(\frac{\partial L}{\partial Q1}\right) = \quad A_1 . T - B_1Q_1 - P_1 - P3 \qquad (10)$$

$$\text{and} \quad \left(\frac{\partial L}{\partial Q_1}\right) = 0 \quad \text{if} \quad Q_1 \neq 0 \quad \ldots\ldots (10)$$

$$\left(\frac{\partial L}{\partial Q2}\right) = \quad A_2 - T - B_2Q_2 - P_2 - P_3 \qquad (11)$$

$$\text{and} \quad \left(\frac{\partial L}{\partial Q_2}\right) = 0 \quad \text{if} \quad Q_2 \neq 0 \quad \ldots\ldots (11)$$

$$\left(\frac{\partial L}{\partial P_1}\right) = Q_{1max} - Q_1 \quad \ldots\ldots (12)$$

$$\text{and} \quad \left(\frac{\partial L}{\partial P_1}\right) = 0 \quad \text{if} \quad P_1 \neq 0 \quad \ldots\ldots (12)$$

$$\left(\frac{\partial L}{\partial P_2}\right) = Q_{2max} - Q_2 \quad \ldots\ldots (13)$$

$$\text{and} \quad \left(\frac{\partial L}{\partial P_2}\right) = 0 \quad \text{if} \quad P_2 \neq 0 \quad \ldots\ldots (13)$$

$$\left(\frac{\partial L}{\partial P_3}\right) = Q_{total} - Q_1 - Q_2 \quad \ldots\ldots (14)$$

$$\text{and} \quad \left(\frac{\partial L}{\partial P_3}\right) = 0 \quad \text{if} \quad P_3 \neq 0 \quad \ldots\ldots (14)$$

Each of these sets of relationships can be interpreted in a perfectly straightforward economic manner.

Thus from (14) all the available water should be used if the price attached to it is not zero (water in this case does not exist in excess). From (12) and (13) the water reticulation system should be fully utilised, provided that this does not exceed the total supply available. From (10) $P_1 + P_3 + T = A_1 - B_1Q_1$. This means that the effective price for water is the sum of the following three components, P_1 the rate fixed by the installation of the reticulation to user 1; P_3 an overall flat rate relating to the relative total abundance of water, and the exogenous reticulation cost T. The same comment applies in an analogous way to user 2 in equation 11.

Clearly the Lagrangian equation (9), can be extended to as many water users as the catchment area or river system carries and will result in an appropriately extended set of Kuhn-Tucker conditions.

From the model developed an optimal allocation of water to each user in the catchment may be determined together with a price which is in a sense market related. It can also be seen that two different types of information are required to solve the pricing/allocation problem for a catchment area or river system. These are demand schedules generated by the different economic sectors using water in their production processes, i.e., economic data, and physical data describing the catchment or river system. This input data required to operate the model is composed of:

(a) Quantities of water required for tin inelastic zone (cf. Figure A3) i.e., for life support and conservation (or other) purposes as the analyst decides.

(b) The marginal costs over time associated with the supply of a unit of water from source (reservoir, dam or river) to each economic sector considered in the particular problem being analysed.

(c) Demand schedules for each of the economic sectors considered in the problem being analysed. Such schedules must be explicitly specified and point out the characteristics of the water demand over time: for example a year, since variations in climate and seasonal demands for both agriculture and industrial products can temporally influence the demand for water in different economic sectors. So far as household water is concerned the diurnal demand for water for lawn watering and domestic purposes are not uniform. It must be noted that whilst the demand schedules shown in the mathematical description of the model are linear, this is not mandatory, any non-linear demand schedule can be dealt with by the model.

(d) Physical characteristics of the water delivery system. In this regard the DWA&F and the various water boards in South Africa are constrained by the physical characteristics of the country's water system. The most important constraint is of course the actual amount of water available at any particular time. Another constraint is the design of the delivery systems which limits the amount of water supplied to any end-user at a particular time. In addition system losses should be known since these have to be taken into account in attempting to meet the demand for water from any particular economic sector.

Whilst the relationship between the model developed here and the physical characteristics of water discussed in section A3.4.2 may seem obvious, it is considered appropriate to re-emphasise the connection. The physical characteristics of water which impact the modelling process can be gathered together broadly under the following headings: temporal characteristics, quantity characteristics and location characteristics. Considering first the temporal characteristics, these have a marked influence on the demand and supply schedules for water, in particular their shape and position, i.e., where they cut the price and quantity axes. The most obvious changes in these schedules are the seasonal changes. For example in the irrigation sector different seasonal rainfall patterns affect the crop growing water requirements and a concomitant increase or decrease in demand for water is registered. Similarly the quantity of water available for distribution can vary from season to season thus affecting the quantity (Q) parameters of the model. A connection is therefore immediately observed between water's temporal and quantity characteristics. Municipal water requirements are similarly affected by season. In summer for example water for gardens and swimming pools is in greater demand than in winter. Industrial water demand will generally not be affected on a seasonal basis but the supply of water to industry will be, thus the Q parameter in the model is affected as is the supply schedule. Temporal characteristics therefore have a significant influence on the structure of the model for allocating and pricing water.

Whilst the quantity characteristics of water supply and demand have also been commented on above so far as its impact on the model is concerned it must be added that the stochastic nature of South Africa's water supply is important. Where for example the model is used for pricing and allocating water in a single year stochastic aspects of the pricing/allocation are ignored. These problems include such things as variations in the demand and supply of water year-on-year. Stochastic characteristics can be catered for by solving the model many times in sensitivity mode. This is a particularly important exercise in South Africa owing to the arid nature of the country and its drought patterns.

With regard to location characteristics it was indicated that the supply price for water is dependent on many things including pumping costs, and the kind of terrain over which water is transported, etc. Depending on where the supply and demand points for water are then in any particular catchment the model's input data will be affected as will the pricing and allocation regime. The location of different water consumers can also dictate the re-use potential of water and thus the in-stream and off-stream water uses. This last issue together with potential reuse clearly affects the Q parameters of the model.

From the brief discussion above it can be seen that so far as the modelling process is concerned water's physical characteristics are important and they can be seen to be interconnected one with another, justifying the claim that to model a water catchment so that an optimal demand-side pricing and allocation regime is maximised, requires the problem to be approached in a systems context.

At its present stage of development, the model is not able to cope with non-consumptive water uses or long-term pricing. The ravages, risks and rehabilitation associated with drought episodes are likewise not taken into account by the model.

These are areas which are not unimportant, but could not be developed within the time scale of the present project. They are issues which could be given serious consideration for development at a later stage.

APPENDIX B

WATER QUALITY MANAGEMENT

CONTENTS

B1. SUMMARY

"The economists' argument for the need to control polluting activities is based on the idea that the free economy, left on its own, will not allocate resources efficiently because the costs which result from discharges or releases into the environment are faced by society as a whole, rather than by the polluting firm. " (Grigalunas, 1988, p509)

The aim of this chapter is to look at the use of market instruments for pollution control, or, specifically for water pollution, the maintenance of water quality standards. The decision maker must take into account two types of pollution sources requiring inherently different control methods: point-source pollution (e.g. factory effluent pipe) and non-point-source pollution (e.g. leached salts from irrigation). The use of economic instruments is mainly presented in the context of point-source-pollution, since applying these instruments to the generally more complex problem of non-point-source pollution proves rather difficult. In order to be able to assess the complexity of pollution control mechanisms, some fundamental concepts are discussed: the notions of externalities, optimal pollution and "environmental property rights". Pollution can be viewed as an economic (external) cost in the sense that polluters do not bear the full cost burden of their activities. When considering pollution levels, the societal optimal solution is not to eliminate pollution altogether but rather to limit effluent discharges to a level at which the total social benefit is maximised. With regards to "environmental property rights" policy-makers are faced with the question of who actually owns these property rights. Based on these concepts, actual economic instruments that can be used to control pollution are discussed.

As far as point-source pollution is concerned, the main instruments the economist is interested in are those that induce a given polluter to internalise externalities. There are a number of ways in which this can be brought about: namely through charges, artificially created 'pollution markets', liability, enforcement incentives and subsidies. These instruments inevitably display limitations as to their actual applicability and in terms of their economic efficiency; they have, nevertheless, been utilised in a number of countries and it proves useful to evaluate these experiences in terms of benefits and disadvantages of the instruments used.

When dealing with the more difficult problem of controlling non-point sources of pollution, it is not possible to restrict the discussion entirely to the so-called economic approaches but other known policy approaches have to be taken into account. To date, the following broad classes of policy measures have been advanced: command-and-control, voluntarism and economic incentives. Within the measure of command-and-control we can identify two basic classes: design standards and performance standards. Both standards, however, have serious limitations, not only because standards by themselves are difficult to enforce but also because the actual setting of standards proves problematic. Another policy measure, voluntarism, involves inducing voluntary participation in abatement activities through education, technical assistance and moral suasion. Finally, economic incentives form the third, and from an economist's standpoint most appealing measure that can be used in tackling non-point source pollution. These include taxes and fees, subsidies, and tradable permits and bubbles.

This appendix does not offer any immediate solutions but is intended to serve as a tool-box for economic instruments that may be used in approaching water quality management in South Africa.

B2. BASIC CONCEPTS

This section introduces basic concepts which are necessary as a background for assessing economic instruments in the control of point- and non-point-source pollution: externalities, optimal pollution and "environmental property rights".

B2.1 POLLUTION AS AN EXTERNALITY

In economic terms, pollution is considered a physical effect of waste on the environment and a human reaction to that physical effect. Physical effects may be auditory, chemical or biological (health etc.) while human reaction is usually some expression of concern or stress towards the physical effect. In short, the human reaction indicates a loss of welfare to the individual concerned.

Consider how such economic pollution may come about. A classic example is that of an industrial plant, upstream of a fish stock used by recreational and commercial anglers, which discharges toxic effluent into the streamflow. Since the angler's activities are negatively affected we may think of the industrial firm as imposing a cost on them. If the firm is not obliged to compensate the fishermen this cost is external to the firms production process, but is nevertheless a cost associated with their activity. Such costs are known as *external costs* or *negative externalities*.

It is, of course, possible that a firm's production process might create benefits for which the firm is unrewarded. An example might be a timber firm plantation which enhances the recreation of the local residents. Such externalities are termed *external benefits* or *positive externalities.*

From the perspective of the policy maker, however, it is external costs which are of greatest concern as they encompass two dimensions: i) some agent imposes a welfare loss on another agent and ii) the fact that this loss is uncompensated (and neither does the offending agent have any incentive to compensate it). There are essentially two ways in which the problem of negative externalities may be addressed.

B2.1.1 Regulatory Legislation

Authorities may act to prevent one agent imposing a welfare loss on another. Typically such policies are reflected in regulatory legislation aimed at reducing or even eliminating the external costs associated with pollution discharges. In practice these policies have tended to mandate the use of certain pollution control technologies or directly set limits on the discharges allowed to individual firms. Ideally an externality elimination policy would reduce discharges to zero. While such a goal is often considered to be worthy, particularly by those espousing an "eco-preservationist" world view, it remains a basic fact of thermodynamics that economic activity cannot occur without the generation of waste. Thus, *to eliminate pollution altogether will require the cessation of economic activity.* This unacceptable extreme points to the existence of a societal trade-off in which the external costs of pollution are offset against the gains from economic activity. Regulatory legislation will be discussed further when considering non-point-source pollution in section B3. of this appendix.

B2.1.2 Internalisation of Externalities

Another approach to resolving the externality problem, which recognises this trade-off, is commonly termed *internalising the externality.* Rather than seeking to explicitly eliminate the externality, policy makers seek to correct the fact that, i) externality creating firms do not incur the full cost of their actions themselves and; ii) sufferers of the external cost remain uncompensated.

Various methods by which externalities may be internalised will be considered in section B3. and will be illustrated in terms of their actual applications.

B2.2 OPTIMAL LEVEL OF POLLUTION

We have already noted that economic activity necessarily generates waste, often in the form of pollution. Since the cessation of economic activity is an unacceptable remedy to pollution, there must be some "acceptable" level of pollution. There are essentially two views on what constitutes "acceptable" pollution, each rooted in a basic world view.

The first view is essentially a physical perspective. It recognises the ability of most natural systems to assimilate a certain level of physical contamination without being degraded. Contamination levels beyond the assimilative capacity of the systems are deemed unacceptable. In the case of particularly sensitive systems acceptable pollution levels, in this sense, may be nearly zero with concomitant implications for economic activity.

The second perspective hinges around the concept of economically optimal levels of pollution. In essence, the view maintains that pollution levels may be increased as long as the incremental value, to society, of the increased pollution (ie the increased benefit derived from the pollution creating activity) exceeds the incremental cost imposed on society by expanding the activity.

Obviously, this could easily result in pollution discharges above the assimilative capacity of the system with long term destructive consequences to the environment. These long term environmental costs are part of the externality cost against which the benefits of increased pollution levels must be weighed. So this economic perspective does not in any sense preclude long term "sustainable development". It does, however, embody the notion that there is a single optimal level of pollution generating activity at which the total net benefit to society (ie economic benefits of pollution generating activity - pollution costs) is maximised. What is this level?

This economic definition is illustrated with the aid of Figure B1. The polluter's activity level is given on the horizontal axis. Marginal or incremental costs and benefits are shown on the vertical axis.

The MNPB or "marginal net private benefit" curve gives the marginal benefit that accrues to the polluter at each activity level ie. the extra net economic benefit from changing the level of economic activity by one unit. The curve is falling because polluters generally receive a diminishing incremental economic return for increases in activity.

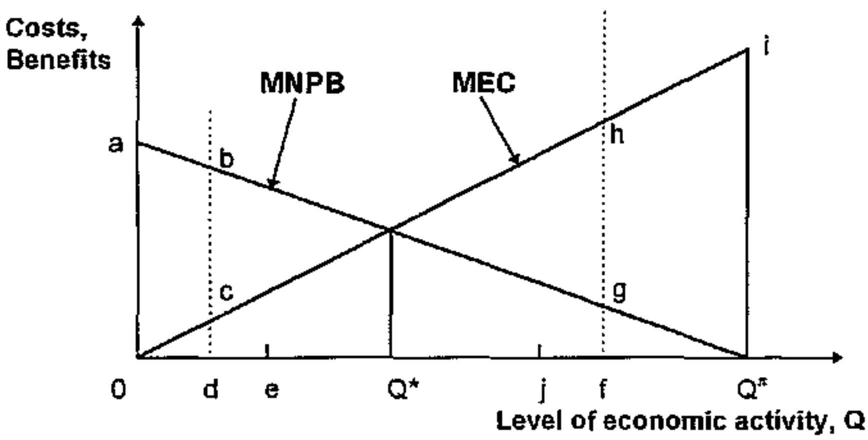

FIGURE B1: OPTIMUM POLLUTION BY BARGAINING

The MEC or "marginal externality cost" curve gives the value of the extra damage done by pollution arising from an additional unit of economic activity. It is depicted as rising since the extra damage usually increases with additional pollution, although other shapes are possible.

For economic activity levels where the MNPB curve lies above the MEC curve, the benefits derived from increasing economic activity by one unit exceed the costs incurred by the action. The reverse applies for economic activity levels above Q^* where the MEC curve lies above the MNPB curve. Consequently, at economic activity levels below Q^*, total social benefit can be increased by expanding the level of economic activity and by decreasing activity for levels above Q^*. Q^*, therefore, represents the economic activity level at which total social benefit is maximised. Accordingly, the pollution associated with this level of activity is the optimal level of pollution.

With this definition of optimal pollution it is clear that the objective of the public policy maker will be to ensure that firms produce output of Q^*. In general, this may be done in two ways.

Firstly, regulators may simply mandate a maximum economic activity level of Q^*. A logical and more practical equivalent of this measure would be to set the maximum allowable pollution at that pollution level associated with output Q^*.

Secondly, we note that the marginal externality cost, at output Q^* is Q^*Y. In the absence of regulation, the firm will not incur this cost itself and will, consequently, produce output of Q^{π}. If the firm were required to pay Q^*Y on each unit of pollution it generated then its optimal activity level would correspond to the socially optimal level viz. Q^*. Thus, an alternative policy which would result in the firm producing at socially optimal levels is one which imposes a pollution tax on the firm. More will be said about these two basic approaches to internalising externalities in subsequent sections. But first we must consider some important issues related to property rights.

B2.3 PROPERTY RIGHTS

As a prelude to a practical discussion of various cost internalisation policies, we shall briefly discuss the role of property rights in allocating the burden of external costs.

In the economist's lexicon a property right refers to the right to use a resource. "This might mean the right to cultivate crops on land that is owned, the right to use one's own house, and the right to use the natural environment in a particular way." (Pearce and Turner 1990).

The question of property rights assumes importance in practical pollution control policy. The fundamental question here is whether firms have the right to pollute or whether the remaining members of society have the right to a pollution free environment.

In the former case, a firm creating an external cost through polluting activity is merely exercising its right to use the natural environment as a receptacle for pollution and there seems to be little basis for imposing on it any form of penalty or regulation for doing so.

On the other hand, the latter case would prohibit the firm from undertaking pollution creating activity unless it were prepared to fully compensate all affected parties.

The assignment of property rights to the environment is clearly a thorny social and ethical issue. Pezzey (1988) has defined two principles, to be used by policy makers, which incorporate slightly differing views of the underlying environmental property rights.

The first of these, known as the Standard Polluter Pays Principle (Standard PPP), is generally applied by most western industrialised countries. This requires polluters to pay for controlling effluent down to the optimal load, but not for environmental damage caused by the optimal effluent load. De facto, therefore, it grants polluters a property right to discharge the optimal level of effluent free of charge. Firms do not, however, have the right to discharge more than the optimal effluent load and are responsible for any costs associated either with additional discharges or abating discharges to the optimal level.

The second principle is less benevolent towards firms in that it asserts that they have no pollution rights at all. Under the Extended Polluter Pays Principle (Extended PPP), polluters are required to pay both the cost of optimal effluent control and the cost of pollution damage done by the remaining optimal effluent.

As Pezzey (1988) argues, these principles are not always distinct in practice:

"It is not suggested that this distinction can be applied rigidly in practice. It may not always be obvious what is the difference between a control cost, a prevention cost, a clean-up cost and a damage cost. Much will depend on where the boundary between 'firm' and 'environment' is drawn. Neither is there much reason to expect any country to adhere rigidly to either the Standard or the Extended PPP in all its pollution policies, nor to shift gradually from one version of PPP to another; both are allowed within the PPP as a whole. What can be said, as a very broad generalisation, is that at the moment most industrialised countries, .. in most cases, apply the Standard PPP rather than the Extended PPP...

There are no good reasons for expecting the current position to change much; industry can generally be expected to oppose any general application of the extended PPP..."

There appears to be good reason, therefore, to assert that the Standard PPP is the appropriate property right framework for South African environmental regulators.

B3. POLLUTION CONTROL METHODS FOR POINT-SOURCE POLLUTION

As mentioned in section B2.1.2 of this appendix, one approach open to decision-makers when faced with externalities is termed 'internalisation of externalities'. Because externality creating firms do not bear the full consequences of their actions they invariably create higher costs than they would if these costs were charged to them. For example, a firm polluting a river will have little incentive to spend money on abatement technology if it can discharge effluent at no cost. However, if the firm were required to compensate fishermen for their losses then there would be some level of discharge at which the cost of compensating fishermen would exceed the abatement cost, and the firm would cut back its effluent discharge. Thus, in being obliged to bear, or at least share, the burden of an external cost the firm's actions are altered and the cost is said to be *internalised*. When costs are internalised the externality problem is resolved by finding an acceptable balance between the cost of reducing an external cost and the external cost itself, not by wholly eliminating the external cost.

The internalisation approach is deeply rooted in the foundations of the market. External costs may be viewed as a distortion of prices which result in firms making decisions on the basis of costs which are too low. By internalising external costs these *market failures* or distorted prices are corrected and once again the market mechanism, of individuals working towards their own ends, will result in the overall maximisation of social welfare. A number of practical tools fall under this approach and they will be discussed in turn below. Namely they are: distributive charges, granted tradable consents and subsidies.

We are now ready to describe a number of policy instruments that may be used to control environmental pollution through what we have previously termed the method of internalising externalities. We will evaluate these methods by describing them in terms of their theoretical implications, discuss the benefits and disadvantages and then look at experiences and particular implementations of these methods in other countries.

The methods described in this section are the ones that have been commonly used for point-source pollution control, ie. for pollution triggered by an identifiable agent and which is relatively easy to trace. There are certainly some overlaps when dealing with point-source and non-point-source pollution in terms of the methods used to control them: when, for instance, dealing with one spatial incident of pollution, methods for point- and non-point source pollution can be used and some instruments that are traditionally used for controlling point-source pollution may also find limited or modified application when dealing with non-point-source pollution. For sake of simplicity, however, we have chosen to discuss these instruments separately.

B3.1 DISTRIBUTIVE CHARGES

Recall in our earlier discussion of optimal pollution levels that we concluded by remarking that public regulators could induce firms to produce at the socially optimal output level either by directly regulating the total discharge to the optimal level or by imposing a tax on each unit of the polluter's discharge. This latter approach is the basis of the distributive charge.

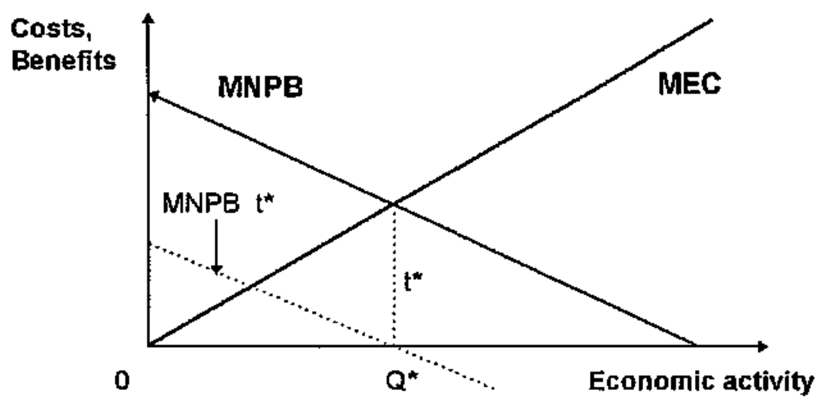

FIGURE B2: THE OPTIMAL POLLUTION TAX

Figure B2 is essentially the same as Figure B1 except that now a tax t^* is imposed on the polluters. It must be emphasised that this tax is a fixed amount applied to every unit of discharge.

THE BENEFITS

The effect of such a tax is to shift a firm's marginal net private benefit curve (MNPB: the extra benefit it derives from each additional unit of activity) downwards by the amount of the tax.

Since the firm is not directly faced with the marginal external cost of pollution it will continue to expand it's activities until the marginal net private benefit from producing more is zero. Without the tax we saw that this resulted in output level Q^P, a level above the optimal Q^*. With a tax, and the new MNPB curve, this level is reduced. If the tax is optimally chosen, the firm's behaviour will be modified in such a way that the optimal level of activity results. Such an optimal tax, depicted as t^* in the diagram, is often called a Pigouvian tax (named for the economist A.C. Pigou who first discussed externalities).

The imposition of a Pigouvian tax is the first element of a distributive charging scheme. The second element of the scheme involves returning the collected tax revenue to the polluters in the form of subsidies for new pollution control equipment. This ensures that Standard PPP applies. If the revenue collected is not returned to the firms, Extended PPP is effectively being applied and the overall scheme is then known as an *incentive charge*.

ASSOCIATED DISADVANTAGES

There are a number of difficulties that have to be faced if a system of distributive charging is to be introduced.

— To calculate proper charge and subsidy rates proves difficult, because if these are incorrect they may not result in the full required reduction in the effluent discharge. In order to calculate these rates, regulators essentially require knowledge of the external costs of pollution and the net benefits which a firm reaps from its polluting activity. This in turn implies the application of some form of *environmental accounting* which ideally requires knowledge of the firm's profit function. In the face of such exacting data requirements it is unlikely that the optimal tax could be computed. Nevertheless, approximations can be made which would move the economy towards an optimum point. In addition, regulated discharge levels can be retained to guard against errors which result in environmentally prudent effluent ceilings being surpassed.

— It proves to be administratively difficult to determine the proportion of a firm's new capital investment that is attributable to effluent control.

— Industry may be forced to spend less on its production activities and more on pollution control, than it does under the existing regulatory framework. While this is the aim of the charge, in the absence of clear evidence that existing emission levels are too high, it is likely to generate considerable political opposition from industry. ((consider Hahn, paper 19))

— The costs of having to administer the subsidy scheme may be very high, although this might be offset by a reduction in the administrative costs of enforcing direct controls.
— A further difficulty is posed by industrial growth or decline. If charges for new sources of pollutants are too low, pollution may increase too much; if they are too high, industrial development may be inhibited. In addition, growth and decay can cause problems for the financing of a distributive scheme: as effluent is reduced, either rates will have to be adjusted so that a falling charge revenue remains adequate to finance subsidy expenditure, or some subsidies will have to be paid in advance of charge revenue.
— Closely related to the previous problem is the problem of inflation. In an inflationary environment, charge rates will have to be continually reviewed so as to maintain their real value.
— Finally, government reluctance to allow the dedication or "hypothecation" of revenue for specific expenditure purposes - that is to handle such revenues and expenditures outside the normal government budget - may make it difficult to undertake the recycling of charges into subsidies that is required by distributive charging.

SOLUTIONS

Despite these difficulties, it has proved possible to set up workable rules for the use of distributive charging, although these usually entail some measure of distortion from the theoretically desirable position because:

— theoretically both operating and capital costs of new and existing emission control equipment should be subsidised. In practice, however, existing equipment is rarely subsidised while it is often difficult to distinguish between pollution control costs and ordinary investment, especially with newer clean technologies;

— it is sometimes impractical to include operating costs in the subsidy scheme, and excluding them biases subsidies in favour of capital intensive emission controls;

— in order to avoid excessive administrative expenses it may be necessary to exclude smaller polluters from the scheme.

B3.1.1 The Application of Distributive Charges

Charge systems of one kind or another have been used in a number of countries with varying degrees of success.

Country	Type of Charge			
	Effluent	User	Product	Admin
Australia	x	x		
Belgium		x		
Canada		x		
Denmark		x	x	x
Finland		x	x	x
France	x	x	x	
Germany	x	x	x	
Italy	x	x	x	
Japan				
Netherlands	x	x	x	x
Norway		x	x	x
Sweden		x	x	x
Switzerland		x		x
United Kingdom		x		x
United States		x		

TABLE B1: USE OF CHARGE SYSTEMS BY COUNTRY

Table B1 above identifies those nations which have used some form of charge system in the control of water quality. In the three sections which follow a number of these policies' implementations are described.

B3.1.1.1 Effluent Charges

To recap, effluent charges are a form of pollution tax paid on each unit of effluent discharged by a polluting firm. Practical implementations of such charges must account not only for the quantity of effluent being discharged but also its quality or environmental toxicity. Attempts have been made in Australia, France, Germany, Italy and the Netherlands to apply such charges. No details concerning the Australian system are available. As far as can be established, all these systems were still operational at the time of writing.

FRANCE

Description

The French charge system, implemented in 1969, is not strictly a "control" instrument in that its function is purely revenue raising. Revenues are used to finance the six "Agencies Financieres de Bassin", independently financed public authorities which dispense financial aid to local authorities and industry for the purposes of constructing treatment plants and other infrastructure projects dedicated to water supply and quality maintenance.

The pollution charges themselves are but one component of a two prong charge system, the other component being a consumption charge on surface and ground water supplies. Pollution charges are levied against anyone who pollutes sea or fresh water and generally include the following categories of discharge: suspended substances, oxidisable matter, soluble salts, inhibitory matter, organic/ammonia nitrogen and phosphorus.

Charge rates for households are calculated each year by municipalities while those for other sources are charged on the basis of a flat rate estimate or by actual measurement. Rates vary by agency and are chosen on the criterion of budget neutrality rather than on the basis of an environmental cost estimate. Total revenues raised under the charge program in 1986 were US $274m.

Evaluation

Two features of this programme diminish its incentive effect. Firstly, pollution charges are set too low to have much of an impact on firms. Secondly, it is estimated (OECD 1989) that the investment aid provided to firms through the agencies offsets the abatement costs by about 12%. It appears, however, that the geographic bias of agency investment has led to firms in specific areas intensifying their abatement efforts - ie the main effect of the program appears to be subsidy rather than charge induced.

Industry has strongly opposed raising the charge rates with the result that rates have decreased in real terms over the past few years. It thus seems unlikely that charges will be increased to the point where they have an incentive effect. This resistance is rooted in a deeper problem succinctly described in the OECD report (1989): " Polluters in France are willing to carry out anti-pollution measures as long as they receive financial aid in return. However, polluters do not agree to higher charge levels which are the source of financial aid. Since, in fact progress in abatement efforts generally implies increasing marginal costs, a deadlock might occur, preventing a speedup of pollution programmes.".

The system would appear to have been partially effective in environmental terms. Organic pollution, which is admittedly the easiest to abate, has been significantly reduced while other substances require further effort. What is not clear, is whether these improvements are a result of the charge system or the direct controls that go along with it. Perhaps the combination of direct standards regulation and the charge system providing further means and incentives to meet them are what has been effective.

Finally, in terms of efficiency it is clear that this charge system is not economically efficient in the sense that it results in the lowest cost to society. It is, however, a simple and efficient system to administrate and provides some degree of adherence to the polluter pays principle.

GERMANY

Description

Introduced in 1976 the German water pollution charge is the only known such charge with a clearly stated incentive purpose. Nevertheless, it runs alongside a command and control system of direct regulations. Formulation of water quality and pollution abatement standards is the preserve of the Federal government. Administration of the system is performed at Lander (state) level.

Five types of pollutants make up the charge base: organic settling substances, oxidising substances, mercury, cadmium and substances toxic to fish. Only discharges into open waters are liable.

Firms are charged for discharge at rates which have risen from US $6.55 to US$22 per unit of discharge over 1981 to 1986 period, rates which have netted about US $154m annually. Charges are levied on the basis of the amount that firms will pollute if they adhere to federal minimum emission standards. It is interesting to note the way in which incentives are linked to these standards. Compliance with the standards results in a 50% discount for the firm. Should a firm be able to prove that its emission are less than 75% of the standards, the charge is calculated according to the actual discharge. For firms planning the installation of abatement equipment, which will result in a discharge reduction of at least 20%, there is a three year exemption from charges. A special clause exists to exempt firms who expect "considerable adverse economic effects" if payment is made.

Revenues collected from the charges are used by the Lander to cover administrative costs and to provide financial assistance to public and private abatement efforts. However, no budget neutrality requirement exists.

Evaluation

Despite a general improvement in German water quality since the introduction of the incentive charges, it is difficult to assess the efficacy of the charges in isolation from the accompanying direct regulation. Total revenues collected under the German scheme are considerably lower than those in France and unit charges are quite small relative to abatement costs. This suggests that the charge levels have not been set high enough to significantly affect industry's behaviour. This, in turn, may simply be a political expedient in the face of vigorous industry opposition to the imposition of a charge system. Having lost this battle, industry became more concerned with the actual charge levels and implementation details (Hahn 1989). With the focus of the debate thus shifted, the federal government can concentrate on gradually increasing charges to the point where they do have incentive effects. Nevertheless, it should be reiterated that the policy as a whole has had a clear beneficial impact on the environment and that public and private enterprises have responded to the system's incentives.

Administratively, the system has proved to be quite inefficient with more than half the revenues being spent on administration costs. More impressively, the Council of Experts on Environmental Questions has estimated that the system achieves environmental goals about two thirds of the cost of a uniform standard policy. If this estimate is accurate it is certainly very encouraging support for the theoretical arguments behind effluent charges.

Finally, the policy does adhere to the "polluter pays principle". By discarding the French emphasis on using revenues to subsidise firms the burden of payment is falling on industry. Unfortunately though, the charges are currently set at levels well below the true cost of the environmental damage being wrought by the discharges.

ITALY

Description

The Italian system, operating since 1976, is also a combination of charges and direct regulations. Two charges are applied: a user charge to raise revenue for the sewer system and treatment facilities and an incentive effluent charge to attain a set of pollution standards.

The effluent charge applies to direct discharges into open waters and is based on the volume, quality and treatment costs of the discharge for individual plants. These are usually assumed values since very little monitoring occurs. An incentive scale based on the standards defines the charges. Firms in violation of the standards pay 9 times as much as those in compliance.

Evaluation

Very little data is available to facilitate evaluation of this system. It has been pointed out (OECD 1989) that this system essentially only differentiates between compliance and non-compliance and is therefore not a true effluent charge but more akin to a non-compliance fee.

NETHERLANDS

Description

The Netherlands have had a charge system in place since 1969 and it is widely regarded as the world's best administered system. The system is a complex one comprising a mixture of user/effluent charges and direct regulations. In terms of its institutional context the system is very similar to the French one in that its primary purpose is to raise revenues for financing projects that will improve water quality. The Dutch water boards are responsible for maintaining balanced budgets and this determines the level of the charges.

Charges are levied on biodegradable matter, suspendable solids, toxic substances and heavy metals. Households and small firms pay a standard charge, medium firms pay according to a table with unit rates for different industries and large firms are monitored. In all cases, reduction of the rate is possible if pre-treatment of effluent takes place.

The most significant feature of the Dutch system is that, unlike other systems, its effluent charges are relatively high. This fact, has resulted in their having an incentive effect over and above the intended revenue generating effect.

Evaluation

The system clearly appears to have been effective. Between 1969 and 1975 water pollution decreased 50% with a further 20% fall up to 1980. Another 10% reduction was estimated to 1986 (Bressers 1983). It was concluded, in this study, that the abatement was a direct result of increasing and anticipated increases in charge rates.

Administrative costs of the system run to only 4-5% of the revenues collected and in this sense it is highly efficient. Economic efficiency, in terms of reducing the overall cost of reaching pollution targets, is only moderate however. This is because, except in the case of large firms, a clear link between charges and discharge is absent in the system. Certainly, this remains an avenue for improvement in the Dutch approach.

Conclusions

In reviewing the effluent charge systems we have discussed, one key point is immediately obvious. Nowhere do charges operate alone, all existing systems have linked effluent charges to a regulatory permit system. Indeed, in most cases the primary goal of the program has not been, as economic theory would urge, to create incentives but rather to raise revenue for abatement subsidies. Thus, the central observation from experience is that charges can be effective tools for raising earmarked revenues. In this we can point to overall success, environmental quality has improved.

Nevertheless, we can also attribute some degree of economic success to them. Incentive impacts have been observed, particularly in the Germany and the Netherlands. German experience indicates that incentive effects may be achieved by the expectation of future increases in fees. Dutch experience confirms that actual fees can lead to abatement actions by firms. The remaining countries illustrate the incentive destroying effects of setting fees too low. Finally, and perhaps most importantly, they have gained acceptability as a means of environmental regulation.

B3.1.1.2 User Charges

User charges are payment for the costs of collective or public treatment of effluents. As such, it is debatable whether user charges are actually an economic policy instrument or merely a payment for abatement services. Accordingly, we will devote very little time to this particular instrument.

Use of such schemes is widespread in the OECD although details differ from country to country. Differences in the schemes usually occur in the charge calculation and the waste producers who are targeted. Most countries charge a flat rate for water treatment with, in many cases, an additional variable charge based on volume and/or concentration of the discharge. Usually, both households and firms pay such charges although a number of countries treat the two groups differently. A taste of such schemes is provided in our discussion of the American and Swedish cases.

SWEDEN

Sweden levies a charge for the treatment of all sewage generated by firms and households. The charge comes in two parts: a basic flat fee and a variable volume based charge which accounts for the majority of the bill. All charges are administered at the municipal level and vary by region. Particularly, some municipalities subsidise households by imposing a higher fee on industry.

The charge has been successful in that a growing number of sewage producers are connected to a well run and expanding treatment system. Some incentive effect has been noted in that industrial producers have decreased the volume of sewage generated. However, a concomitant rise in the pollution load has accompanied this fall, suggesting a weakness in the system.

UNITED STATES

The American charge is also administered at the municipal level and, for households, is almost identical to the Swedish system. For firms however, an additional pollution strength charge is levied.

The intent of the strength charge is to charge a high fee for more contaminated waste water. Practically, however, it is very difficult to monitor individual firms so a group-based bill is levied on groups or sectors of firms. Only in the case of large dischargers are individual strength fees applied. This means that these charges, in general, have little or no incentive effect.

A further limitation of the US system is that, often, water use and discharge fees are billed together. This has the result, in cases where incentive reductions occur, of reducing water use but not necessarily pollution discharge, resulting in higher pollution loads in the discharge.

CONCLUSIONS

Most importantly, it should again be stressed that user charges are not designed to be incentive charges. Rather, they are primarily revenue raising fees used to cover the cost of waste water treatment. In this respect they have been largely successful. By lowering the costs of treatment, through publicly coordinated economies of scale, they have induced more dischargers to connect to sewer systems and in so doing have increased the overall level of water treatment.

Attempts to link actual discharge strengths and volumes with the charges, while desirable, are not usually feasible due to the high monitoring and administration costs. Thus, a true incentive use of such charges appears unlikely.

B3.1.1.3 Product Charges

Product charges, by taxing a pollution causing product in the manufacturing or consumption phases, are a primary economic means of controlling non-point source pollution. By their very nature they are not applied to specific resources but rather to products which may pollute specific resources. International experience of product charges applied to water polluting products includes taxes on pesticides, batteries, lubricant oils and fertilizers. Table B2 below gives a fuller account of the international use of product charges.

Country	Product	Purpose	Start
Finland	Non-returnable containers	I	1976
	Lubricant oils	RR	1987
	Crude oil and products	RR	1972
France	Lubricant oils	RR	1981
Germany	Lubricant oils	RR	1969
Italy	Lubricant oils	RR	1985
	Plastic bags	I	1988
Netherlands	Lubricant oils	RR	1979
	Fossil fuels	RR	1972
	Car fuels	RR	1981
Norway	Non-returnable containers	I	1981
	Fertilisers	RR	n a
	Pesticides	RR	n a
	Mineral oils	I	na
Sweden	Oil products	RR	1984
	Fertilisers	RR;I	1984
	Pesticides	RR;I	1984
	Hg/Cd batteries	RR;I	1987
	Beverage containers	RR;I	1973
United States	Feedstock	RR	1981

RR: Revenue Raising; I: Incentive Source: OECD (1989)

TABLE B2: INTERNATIONAL USE OF PRODUCT CHARGES

We shall briefly consider some of these product charge systems, focusing on those countries which have applied incentive charges to water polluting products.

GERMANY

Although primarily revenue raising in nature, the German charge on lubricant oils is widely acclaimed as a highly successful piece of environmental legislation. Regulations, requiring better separation of different types of waste oil for improved collection and recycling/treatment, and charges have been in place since 1969. Importers and manufacturers of lubricating oil have been required to pay a charge of $110 for each tonne of lubricant oil. These revenues are used to subsidise the implementation of the regulations are eventually designed to be phased out (actually planned for 1989).

Aside from the considerable revenue generation, the combined effect of the regulations and the charge has been highly effective. Waste oil disposal in a manner contrary to regulations has fallen from 92 000 tonnes in 1969 to 5000 tonnes in 1981 and further decreases are expected.

NORWAY

In an effort to reduce atmospheric sulphur emissions, Norway has imposed a two part charge on mineral oil. The first part is a general tax on mineral and is of little interest to this discussion. The second part of the charge is based on the sulphur content of the fuel. The intention is to induce industries to shift towards lower sulphur fuels. Despite a 1988 increase in the charge however, it is felt that the charge has been too low to induce the desired shift.

SWEDEN

Fertilizers

The fertilizer product charge adds approximately 5% to the price of fertilizer and is intended to have both a revenue generating and an incentive effect. The charge is levied on the basis of the phosphorous and nitrogen content of the fertilizer. Revenues raised through the product charge are earmarked for use in environmental programs such as research and development in agriculture and forestry. The incentive effect of the charge appears to have been very low and there is talk of raising the charge. However, demand for fertilizer is very unresponsive to price and it appears that a charge nearly equal to price will be required for there to be any significant effect.

Pesticides

The charge on pesticides is very similar to that on fertilizers with earmarked revenues going towards the same programs. The program has experimented with both weight based charges and crop area based charges. In both cases the charges appear to have been too low for any significant effect.

Batteries

A charge on batteries containing mercury or cadmium was introduced in 1987 with an incentive goal of reducing such emissions from batteries by 75% before 1988. At the time of writing, the impact of this program was not known.

CONCLUSIONS

The following observations about product charges are noteworthy:

a) In general they have had very little incentive impact. This can largely be attributed to the generally low charge levels.

b) In many cases, particularly for lubricants, the product charge system has been effective in raising revenues. Where this revenue has been used to subsidise standards compliance the resulting environmental improvements have been substantial. The German lubricant oil charge is particularly instructive in this case.

c) When linked to existing tax and excise systems the product charges are usually quite administratively efficient.

d) Product charges are generally compatible with the polluter pays principle since only the users of the given product are actually paying the pollution charge. Again, however, the question of low charge rates leads to the observation that these users are not paying the full cost of their activity, indicating only partial fulfilment of the polluter pays principle.

B3.1.1.4 Administrative Charges

In some countries the license or registration/control fees that accompany a regulatory control system can have an incentive effect. In general, however, these are not economic instruments.

Although most OECD countries make use of administrative charges there is very little indication of their being used as incentive charges. Indeed their is no evidence to suggest that they have had an incentive effect in reducing emissions. For the most part, these charges are a simple revenue raising device to cover the administrative costs of licensing and control in a regulatory system.

B3.1.2 Future developments

B3.1.2.1 Sewerage and Sewage Disposal

The sewerage and sewage disposal sector deals with the disposal and subsequent treatment of effluents through sewerage systems.

Users of public sewers are of two types: domestic and industrial.

B3.1.2.1.1 Domestic-type Sewage

It is very difficult and expensive to measure waste volumes for every domestic user of sewers and sewage treatment.

Moreover, on the one hand, since for most residential users "the quality and composition of waste water is approximately the same, the major differences are in volumes and thus the input of water to the premises is usually a satisfactory proxy for the sewage generated." (OECD, 1987). On the other hand, because domestic sewage disposal is normally linked to public water supply charges there is no information about the effects of variations in waste charges on sewage flows and sewage strengths.

Given the above, it seems that a sewerage and sewage disposal 'supplement' linked to the structure of the public water supply tariff is the most appropriate solution to the problem of sewerage and sewage disposal charging. For financial reasons this 'supplement' should take the form of a user charge, i.e. a charge which by its revenues covers all expenditures related to the collection and treatment of domestic-type sewage.

B3.1.2.1.2 Industrial-type Sewage

As far as industrial users of public sewers are concerned there is room for the introduction of marginal cost pricing.

"The chemical qualities of water-borne wastes may differ enormously from one discharger to the next and in this situation a charging system reflecting differentiated charges for specialised treatment of different characteristics of wastes may well be justified." (OECD, 1987).

Moreover the results of four US investigations of the effects of variations in waste charges on sewage flows and sewage strengths show that price increases cause significant reductions in the demand for trade effluent treatment services. A summary of these results is given in Table B3 below.

	PRICE ELASTICITY OF	
	Water Demand	**Weight of BOD**
Price of water	-0.60 / -2.20	-0.4 / -2.20
Price of waste strength	-0.30 / -0.80	-0.05 / -0.80

NOTES: Price of waste strength variously measured as surcharge per 100 pounds of BOD and combined surcharge on BOD and suspended solids (BOD = biochemical oxygen demand). Price of water = price of water supply + sewage charge; variously measured as gross and net, where net price obtained by subtracting the value of the free wastes obtained per unit of added water which results from the defined normal levels in the waste-strength charges.

TABLE B3: ESTIMATED PRICE ELASTICITIES IN THE US INDUSTRIAL SEWAGE CHARGES INVESTIGATIONS

[Source: Ethridge, 1970 ; Elliot, 1972 ; Hanemann, 1978 : McLamb, 1978.]

The marginal cost calculation for industrial-type sewage proceeds as follows:

Resource-use costs should involve customer ("one-off" and "continuing"), commodity and capacity costs.

For the calculation of marginal capacity costs the authority should "seek to identify the increased costs were it to be redesigned to take (a) a higher volume (other characteristics of sewage held constant), (b) a higher BOD load (volume and other quality characteristics held constant), etc." (OECD, 1987). In other words, marginal capacity costs should involve marginal capacity costs for "volume-related" facilities, "BOD-related" facilities and any other appropriate "quality characteristic-related" facilities.

Regarding natural resource-depletion costs no costs should be identified. They have already been taken into account under the public supply system (in general users of public sewers are connected to the public supply system).

Damage costs should be taken into account through a "supplement" corresponding more or less to some approximation of the damage imposed to the environment.

This "supplement" should take the form of an effluent charge. Effluent charge systems like any charge system "depend on both volumetric and a number of quality-related elements, known as strength charges (e.g. BOD or COD expressed in mg/l or as a proportion of "permitted" BOD/COD, suspended solids, various heavy metals, toxicity to fish, etc.)." (OECD, 1989b).

Examples of marginal cost pricing for industrial-type sewage are not available. However, examples of incentive based effluent charges will be given in the section related to direct discharges.

As far as transferable water rights are concerned it should be stressed that such a system does not readily apply to sewerage and sewage disposal because important resource-use costs are involved with the sewerage and sewage disposal.

B3.1.2.2 Direct Discharges

This section deals with the direct discharges of effluents to receiving waters.

Direct discharges incur only damage costs.

Direct discharges are made mainly by municipalities, industry and agriculture. Municipalities and industry are primarily point sources of pollution, i.e. they discharge effluents after some form of collection has taken place. Agricultural activities are mainly non-point sources of pollution, i.e. they discharge effluents without proceeding to any collection.

Possibilities of pricing damage costs vary according to the type of pollution source. Point sources are susceptible to direct monitoring, thus effluent charges are applicable. Non-point sources involve many polluters and they are almost impossible to monitor directly. Other environmental charges, that do not require monitoring, are applicable.

B3.1.2.2.1 Point Sources

There are no empirical estimates of price elasticities. There is, however, evidence concerning the incentive effects of direct discharge effluent charges in Germany and The Netherlands.

For The Netherlands, even though the introduction of the effluent charge system was followed by a major investment in treatment systems and a decline in pollution levels, other policy instruments accompanied the introduction of the charge system and it is therefore not possible to separate the cause and effects of each element (United Nations, 1980).

For Germany the incentive effect of the imposition of effluent charges was significant and clear-cut. A brief presentation and evaluation of the German system is given just below.

In 1976 a Federal Act was passed, laying down that the Lander "should, by 1981 levy a uniform national system of waste water charges on the direct discharge of specified effluents into public waters" (OECD, 1987). The charges are levied 'per unit of noxiousness', with five groups of pollutants translated into units of noxiousness: settleable solids, COD, cadmium, mercury and toxicity for fish. Minimum discharge standards compatible with 'commonly accepted technology' was laid down for individual industries and municipalities of different sizes. If direct discharges meet these minima the charges payable are reduced by 50 per cent.

The mere passing of the Act had such a significant effect that by 1981 "10 per cent of waste dischargers were meeting the minimum discharge standards." (OECD, 1989a) "There are also reports from the pulp, electroplating and textile refinement industries of innovative reactions to the Act which induced a general modernization of production processes and therefore enhanced competitiveness as well as large energy savings. Economical washing processes and materials reclamation were frequent examples." (OECD, 1987). Moreover, the Council of Experts on Environmental Questions (quoted in Brown and Johnson, 1983) estimated the economic efficiency of the effluent charge policy to be about one-third cheaper for the polluters as a group than the uniform standard policy. The incentive impacts of the charge may have indeed resulted in a more careful choice of efficient solutions.

The imposition of a direct effluent charge in order to account for the damage costs of direct discharges seems to contribute to the efficient allocation of resources and it should therefore be seriously considered as an alternative to the imposition of standards.

Example of marketable pollution permits

B3.1.2.2.2 Non-point Sources

Due to the non-point character of agricultural production, emissions - such as run off or leaching of nutrients or other active ingredients - as well as emissions of pollutants into water, are almost impossible to control. However, as pollution damage - on site as well as off-site - is mostly caused by the use of certain inputs, mainly fertilisers and pesticides, the easiest way to price the environmental damage is by levying a charge, called a product charge, on those inputs.

Regarding fertilizer charges "studies suggest, that price elasticities are rather low, so that charge rates would have to be very high in order to induce major reduction in the intensity of fertilizer use." (OECD, 1991). Swedish studies report that fertilizer charges have an incentive effect only if they are equal to the sales price of fertilizers (OECD, 1989a).

Fertilizer charges equal to the sales price of fertilizers may cause drastic income losses to farmers. Political considerations will thus determine whether such an incentive fertilizer charge is desirable on social grounds.

Regarding pesticide charges there is no evidence on price elasticities. It is expected, however, that price elasticities are low (OECD, 1989a).

Investigations on price elasticities are thus required in order to determine how high a pesticide charge should be in order to induce reductions in pesticide use.

There are no available examples of incentive use of fertilizer or pesticide charges. Fertilizer and pesticide charges for revenue-raising purposes are used in Sweden and Norway (OECD, 1989a).

Also, there are no examples of marketable pollution permits in the areas of fertilizer and pesticide use.

B3.1.2.3 Modelling Water Quality Characteristics [1]

In this section the model developed thus far for pricing and allocating water is extended to take into account pollution loads carried in the water supply system. The problem that arises is specifying the value function for water in various uses

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1 This section is based on a model developed by GA Veck and DJ Stevens.

where the value depends upon the quantity of pollution contained in the water. When water is polluted its value to various consumers may decrease and this value will depend on the amount of money they, or the water supplier, has to spend on improving the water quality to minimum required hygienic or aesthetic standards. For example, water supplied by municipalities to homes and industry has to be of a certain quality, in the first instance to reduce the risk of disease and in the second to prevent as much as possible the corrosion of plant. This quality is heavily influenced by the ever-increasing quantities of urban, industrial, mining and agricultural effluent which becomes integrated into the water supply system. The principle metropolitan areas are particularly under threat in this regard. The area of supply of the Rand Water Board for example generates more than half the GDP of South Africa from the economic viewpoint therefore the less money that has to be spent on preparing water, quality wise, for its various uses the better. The importance of being able to determine the influence of pollution on the water pricing/allocation problem is therefore of great importance and is illustrated by the fact that the DWA&F have reported that quality of water in time may become a more important factor than quantity so far as availability is concerned[2]. Hence determining the effect of the quality characteristic on the pricing and allocation of water is becoming an increasingly important requirement in the development of economically based water models. Deviations from the minimum requirements would involve a capital outlay to correct resulting economic externalities. The format of the model developed thus far can be readily modified to take into account the adjustments to price and allocation that has to be made to control the pollution in water used for various end-uses.

In this regard it will be recalled from Appendix A (Figure A3) that the model has a demand function for water of the form: Pd=A-Bx,. If a pollution load exists in the supply system this demand function can be modified to (cf., Figures B3 and B4):

$$p_i = a_i - b_i x - \left(\frac{c_i}{x_i}\right) \qquad \text{......................................} (15)$$

Where p_i is the price paid by the water consumer i for x_i units of water, and a_i and b_i are constants. The quantity c_i is the decrement in value of the water demanded due to a fixed amount of dissolved pollutants being contained in the water, and essentially is a parameter which represents the change in water value due to the pollution load.

2 Management of the Water Resources of the Republic of South Africa, op.cit., p. 4.9.

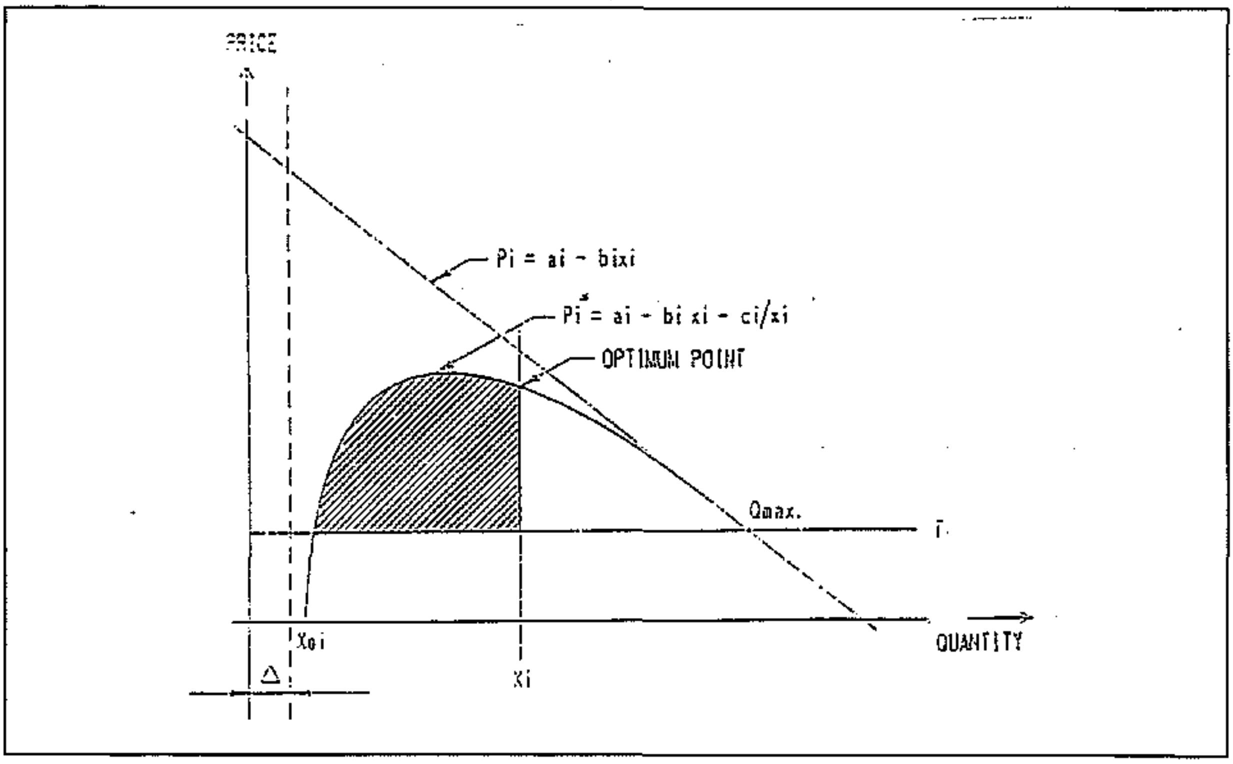

FIGURE B3: SOCIAL WELFARE AND WATER POLLUTION

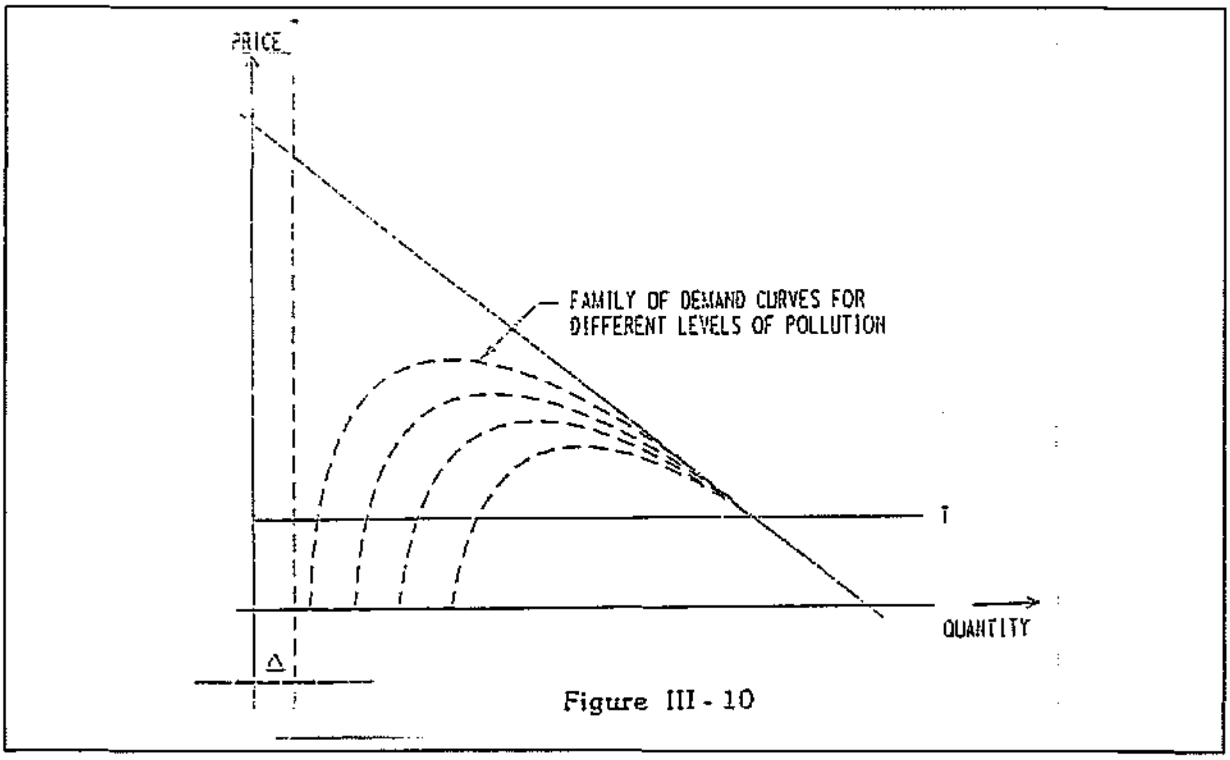

FIGURE B4: DEMAND CURVES AND WATER POLLUTION

Now c_i represents a fixed amount of dissolved pollutants in a fixed quantity of water x_i. Since there is assumed to be a fixed mass of dissolved materials in the water the greater the volume of water under consideration the less will be the affect of these pollutants, c_i is related also to the particular pollution released into the water. Hence for one class of pollution so many cents/litre will be required to reduce the detrimental effects this pollution has on a particular consumer and for another class of pollution a different amount of money may have to be spent to render the water acceptable. Therefore the more expensive it is to decontaminate the polluted water the greater the value of c_i.

Reference is again made to Figures B3 and B4. These Figures show the demand schedule $p_i = a_i - b_i x_i$, the supply schedule for water T, and the adjusted demand schedule for water after taking into account the pollution constraint c_i. Firstly it is seen in Figure B3 that for x_{0i} units of water demanded the price is zero. This in interpreted to mean that the pollution concentration of x_{0i} units of water is such as to make water valueless to the consumer being considered. Secondly for much larger volumes of water the price becomes asymptotically equal to that which obtains for water in the unpolluted state: i.e., p_i is tangential to p_i. Thirdly it can readily be imagined that a whole series of adjusted water demand schedules can exist depending on the number of consumers considered to exist in the analysis and the type and quantity of pollutant contained in the water body under review (cf.,Figure B4).

Returning to Figure B3 the shaded area is an approximation to the social welfare which must be maximised this follows from the discussion previously held in this chapter. The social welfare for this situation is given by:

$$TSW = \int_{x_{0i}}^{X_i} (a_i - b_i u_i - \frac{c_i}{u_i} - T_i)\, dv_i$$

and is a maximum if X_i were to equal Q maximum: u_i is a dummy variable of integration, i.e., if the integration is taken between two constants the dummy variable disappears in the answer.

For n consumers of water the social welfare generated is given by:

$$TSW = \Big[a_i (X_i - x_{0i}) - b_i \frac{(X_i^2 - x_{01}^2)}{2} - c_i \, Log_e \left(\frac{X_i}{x_{0i}}\right) - T_i (X_i - x_{0i}) \Big] \quad \text{.................................. (16)}$$

As in the pricing/allocation problem without pollution the determination of social welfare is dependent on certain constraints, e.g., the total quantity of water available in the system and the amount of water that the delivery system can provide to different end users.

Because the different consumers are competing for the available water the optimal points X_i will generally lie below their respective maxima as shown in Figure B3. The nature of the function to be maximised now contains the seemingly rather innocent logarithmic term which however complicates the solution since it is no longer amenable to the quadratic programming regime used when the quantity, location and temporal constraints of the problem were being analysed. To enable the quality constraint to be analysed therefore an iterative method of solution is necessary. The approach undertaken here is that of the separable programming technique.

In summary, the method relies on the objective function, which depends on a number of variables, being able to be expressed as the sum of a number of funcitons, each of which depends on only one variable. Any continuous function of on variable can be approximated over a range of the independent variable be a sequence of straight line segments. Within the range of a given linear segment, the function is approximately a linear function of that variable.

Because the overall objective function is a sum of such functions, each one of which is approximately linear, the overall function is linear in each of the independent variables, and is thus amenable to maximising or minimising by the simplex linear programming approach. The separable variables approach is partucularly suitable for the objective function given by equation 16, since inspection shows that TSW is already a sum of n functions f_i, each of which depends only upon x_i.

A plot of any of the functions f_i is shown to be concave, and thus an optimum solution to the problem will approximate the true global optimum of (16). The accuracy of the method depends upon the number of linear segments used to approximate any function f_i between x_{0i} and Q_{maxi}, which is the correct range of each independent variable. A set of water allocations and prices are therefore found which on the one hand minimises the cost of suuplying water to various end-users and on the other hand maximises the respective benefits or local social welfare taking account the pollution loads carried in the water supply system.

B3.2 POLLUTION MARKETS

The creation of artificial "pollution markets" is undoubtedly one of the most innovative and debated ideas in the current environmental policy debate. Essentially, the idea is to create institutions whereby actors can buy and sell "pollution rights". To date, there is international experience in two forms of this idea: liability insurance and the more popular marketable permits.

B3.2.1 The Economics of Tradable Permits

As in the case of distributive charges, granted tradeable consents attempt to restrict the total effluent discharge to that produced by the optimal level of economic activity, Q^* in Figure B1 above. However, instead of inducing firms to abate effluent discharges through taxation, authorities directly control the total amount of pollution that may be created. Each firm in the economy (or river catchment, region etc.) must possess a consent (permit) allowing them to discharge a maximum quantity of effluent. The total effluent permitted under all consents will amount to the economically optimal pollution level.

A key feature of these consents is that they be tradeable. Thus, if a modern plant is able to abate its pollution relatively cheaply it may do so and sell the excess "pollution right" to another firm which may find the purchase of a consent cheaper than abatement. In this way, the reduction of pollution to the optimal level is achieved in the most efficient manner - something which also occurs under distributive charging but which does not generally occur if each firm is regulated with a non-tradeable permit as in most current regulation.

A pertinent question related to tradeable consents is how they are initially issued to firms. Two possibilities exist.

Firstly, they may be sold or auctioned to polluters, the resulting revenue entering the general treasury. Under this arrangement polluters are required both to abate their discharges to the consented levels and they must pay for the consent itself ie. the Extended PPP is being applied. Such a consent is known as a *sold tradeable consent* and will not be considered further on the grounds that it is excessive in terms of the Standard PPP desiderata.

The second possibility is to issue consents free of charge to polluters. In this case the Standard PPP applies and the instrument is known as a *granted tradeable consent* - the subject of the remainder of this section.

The significant feature of consents is that they control effluent quantities directly rather than via price adjustments. Because consents operate on maximum permissible discharges, the first difficulty that they cause is that limiting their size - by having authorities buy them up for example - will have little immediate effect in those cases, which may occur quite often, where actual discharges are below those maximum amounts. This means that the introduction of a system of granted tradeable consents is likely to be ineffective in achieving optimal pollution levels unless the limitation in size of the consents is tackled with some vigour.

There are two ways to go about this, unfortunately both facing potential political barriers.

Firstly, at the time of the initial free issue of consents equivalent to existing emission levels, the authorities could announce that after a suitable period - long enough for trading in consents and adjustments in emission control investments to occur - the size of all consents will be scaled down by the same proportion. Although transition costs could be minimised by adopting this approach, the fact that scaling down is to take place could lead to resistance by industry to the introduction of the consents in the first place.

Secondly, this resistance might be removed if the authorities were to buy up the number of consents needed to effect the reduction, but this would grant industry a de facto right to pollution levels greater than the prevailing optimum and so would be unacceptable in terms of the Standard PPP, putting the burden of pollution reduction on the tax payer rather than on the polluter.

A further problem that would be created by the introduction of granted tradeable consents is that unrestricted trade between polluters would probably lead to emission levels increasing in certain geographical regions, even if countrywide emissions are reduced. If such local increases are unacceptable, unrestricted trade cannot be allowed. Thus it will be necessary to make it subject to regional constraints, potentially complicating the system as well as reducing the trading opportunities consent holders.

Another difficulty arises when there is imperfect competition in the market for consents. A market outcome will not be cost-effective if a single large polluter can influence the price of consents substantially, or even create a monopoly.

It is also possible that markets will be thin - that is, that trades in discharge consents will rarely occur so that there is no opportunity for a proper competitive price to be established. The same question also points to the need to address what will happen when a new firm enters the polluter industry - will the non-availability of consents be allowed to be a barrier to entry into the industry.

Finally, there may be a hoarding problem. Firms may get credits for emission reductions and hoard them against possible future tightening of emission targets by the authority. Sometimes, particularly where there is an initial gap between consented and initial pollution, these reductions may be on paper rather than in actual emissions. Hoarding can be tackled either by buying up the hoarded consents, thus breaking the PPP, or by forcing consents closer to actual emissions before starting the trading scheme, which may provoke protests from polluters.

B3.2.2 The Application of Marketable Permits

While under discussion everywhere, the concept has only been extensively implemented in the United States, European countries having tended to prefer a charge system. Some trading of air emission rights has been implemented in Germany but we shall confine ourselves to the American experience.

United States

Three tradable permit systems have been attempted in the US, the most comprehensive relating to air emissions. We discuss each in turn.

Wisconsin Fox River Water Permits

Implemented in 1981, the scheme was designed to control biological oxygen demand (BOD) in the Fox River while allow firms the maximum flexibility to achieve the environmental goals. Regulations were primarily aimed at paper and pulp plants and municipal waste treatment plants.

Firms were issued with five-year permits which defined their initial wasteload allocation. Trading of the permits is limited by location, with firms being divided into clusters (of 6 or 7 firms) so that trading will not increase BOD at two critical points in the river. Any transaction requires modifying or reissuing permits and transfers must be for at least a year. Once the five year period is up firms waive any claim to the rights in their possession. Furthermore, firms are required to justify the need for additional permits on a basis other than a reduction of operating costs.

Perhaps not surprisingly, in view of the restrictions, the scheme has been a failure with only one trade recorded in six years (Hahn 1989). Two reasons for this seem likely. Firstly, the clustering rules have created very thin markets for potential trades. Secondly, the restrictions on trading (Novotny 1986) considerable uncertainty and transactions costs which have discouraged trades.

Another factor influencing the limited participation of local firms is the fact that the scheme was a relatively small part of a wider regulatory structure which specifies treatments and operating rules. Consequently, there is not much to be gained by engaging in trade.

EPA Emissions Trading Program

US efforts to maintain air quality standards are founded on the 1955 Clean Air Act. This act, together with amendments, is fundamentally predicated on the command and control ethic and specifies ambient air quality standards based on the best available technology. To facilitate greater industrial flexibility, the emissions trading program was introduced in 1974.

The program has four distinct policies which were phased in between 1974 and 1979: netting, offset, bubble and emission banking policies, which all apply to emissions of single pollutants. The programme does not allow the trading off of one type of emission against other types. The four policies are linked by a common element: the emission reduction credit (ERC). This credit is essentially a currency which is used in trading among emission points. If an agent decides to control any emission point to a higher degree than necessary to fulfil its legal obligation, it can apply to the control authority for certification of the excess control as an emission reduction credit.

1. Netting or internal trading allows single firms to create new emissions at a plant by reducing emissions from another source at the plant. This allows the firm to avoid the stringent regulations which normally apply to new emission sources.

2. Offsets are emission reduction commitments by existing firms which must be obtained before major new or expanding sources can begin emissions in certain areas. The offset policy applies to so called "non-attainment" areas, regions whose ambient air quality is less than that required by the standards. The offset policy was innovative in that it opened the door to new emissions in non-attainment areas where previously no new sources were permitted in such areas.

3. Bubbles are geographic collections of emission points whose total emissions are regulated. Firms and plants within bubbles may trade ERCs to alter individual source emissions while maintaining the overall bubble emission constant. Such trading applies to existing emissions (unlike netting which only covers new emissions) and may involve either inter- or intra-firm transactions.

4. Banking allows firms to store unused ERCs for future use or sale in the netting, offsets or bubble programmes.

Hahn and Hester (1986) offer the following table in evaluating the emissions trading program:

Activity	Estimated no of internal transactions	Estimated no of external transactions	Estimated cost savings (millions)	Environmental quality impacts
Netting	5000 to 12000	None	\$25 to \$300 in permitting costs \$550 to \$12000 in abatement costs	Probably insignificant
Offsets	1800	200	Probably in \$100s	Probably insignificant
Bubbles	129	2	\$435	Insignificant
Banking	<100	<20	Small	Insignificant

TABLE B4: EMISSIONS TRADING PROGRAMME

It is clear that the majority of activity under the program has taken place in the form of internal or intra-firm transactions. Furthermore, in environmental quality terms, the program has added little to the regulatory program. The major benefit of the program has undoubtedly come in the cost savings to firms attempting to comply with emission standards. These would appear to run into billions of dollars. Tietenberg (1990) also points out that the level of compliance with the basic provisions of the Clean Air Act has increased as the possible means of compliance have expanded. In general evidence suggests, that compared to command-and-control policies, the emissions trading programme can achieve air quality goals at lower costs.

On the minus side Weiss and Palmisano (1985) point out that administrative and transaction costs for the system are high and can have a considerable impact on ERC prices. The emergence of independent ERC brokers gives testimony to these high costs and the burden on regulators.

Dillon Reservoir Trading System

The only documented application of emissions trading to non-point pollution sources, this program was initiated in 1984 at the Dillon Reservoir in Colorado. Under the trading system, point source discharge may be increased when non-point discharge is decreased ie non-point pollution is traded against point sourced pollution. In the Dillon case, four waste water plants were allowed to increase their phosphorous loadings provided they obtained non-point reductions (2 units for every additional unit of point discharge) from other lake polluters. Regulators do not monitor non-point reductions but instead estimate the capacity of installed control equipment prior to granting the trade permit.

Studies (OECD 1989) indicate that considerable cost savings will be achieved, it often being less costly to abate non-point sources than point sources. No further evidence on the results of this scheme are available although it is viewed quite favourably in the literature (Tietenberg 1990, OECD 1989).

CONCLUSIONS

Several observations arise from the discussion above.

> The failure of the Fox River scheme points to potential dangers with tradable permit systems. Firstly, there is the problem of thin markets caused by the clustering provisions. These are analogous to the EPA bubbles and are necessary to ensure that the pollution burden does not accumulate in a spatially biased pattern. Secondly, the administrative costs of tradable permit systems are quite high. Increasing transaction costs by further restrictions on trades can raise the price of permits to the point where it is not economical for firms to participate. Lastly, tradable permits must clearly demarcate property rights to pollute. The uncertainty generated by the five year validity of the Fox River permits undoubtedly contributed to firms' reluctance to enter the scheme.

Both the emissions trading program and the Dillon reservoir scheme illustrate the potential cost savings that tradable permits can engender. As such they are effective in inducing more firms to comply with standards which are an essential prerequisite of permit trading systems. This is compelling reason to turn to such schemes.

Finally it should be noted that tradable permit systems have attracted considerable opposition due to they fact that they effectively legitimise the right to pollute, anathema to those who maintain that a clean environment is a basic right.

B3.2.3 Liability and Liability Insurance

Strict liability and liability insurance for environmental damage is an economic instrument that is receiving some attention in policy formulation circles notably in the United States, Finland, Germany and the Netherlands. By burdening firms with liability or insurance premiums for potential damage it is felt that firms will implement stricter measures to limit their environmental impact.

The most graphic example of such a policy is the much reported case of the Exxon Valdez and its devastating oil spill in Alaska. US courts have already penalised Exxon billions of dollars in clean up and compensation costs and legal action continues. Exxon and other oil companies have responded by paying closer attention to their transportation and other practices (e.g., the introduction of double hulled tankers), signs that liability policies would indeed be environmentally effective.

CERCLA - Comprehensive Environmental Responses, Compensation and Liability Act

In 1980 the Comprehensive Environmental Responses, Compensation and Liability Act (CERCLA) was passed, which together with other US laws pertaining to environmental legislation, establishes strict liability for damages and potentially provides a national framework, resting on the use of financial incentives.

Although the liability established only relates to oil spills or hazardous substance release, a discussion may prove useful since CERCLA provides for immediate assessment of pollution incidents through a computer database which enables policy-makers as well as the polluting party to quickly assess actual damages and the associated costs following an incident. The natural damage provisions in CERCLA is restricted to publicly owned or controlled natural resources and do not include damages to private property.

Under CERCLA and CWA, as amended (Water Quality Act, 1987), the polluter is not only reliable for the clean-up and the reasonable assessment of costs but also for "damages for injury to, destruction of, or loss of natural resources...." (quote in Grigalunas, 1988, p.511)

There are two types of natural resource damage assessment (NRDA) regulations provided for in CERCLA: Type A constitutes the simplified approach, consisting of minimal field observation to be used for minor incidents and Type B presents methods for site specific NRDA, potentially with extensive field observations, for major incidents. The latter type also considers restoration and replacement costs which are to be considered when determining the amount of damage.

This two-tiered system proves very useful when considering the potential assessment costs of the Type B which may be very high and - unless restricted - can easily outstrip the value of the damage.

The primary goal of the act is to encourage distributional equity by compelling the responsible party to pay damage compensation for injuries which incur as a result of their action. With CERCLA a legal framework was created which is similar to a Pigouvian tax on pollution and is considered to potentially be an important new approach for using economic incentives to avoid (point-source) pollution from a wide array of incidents.

Like other assessment methods, the damage assessment in CERCLA is battling to determine the correct value of damages incurred. For simplification, the act provides for establishing measures of damages based on units of release or units of affected area, recognize the need to use average values and approximations rather than site-specific values. The NRDAM/CME (Natural Resource Damage Assessment Model for Coastal and Marine Environments) for Type A is composed of three submodels: the physical fates submodel, the biological effects submodel, and economic damages submodel of which we want to consider the last one. Within the economic damages submodels, damages are measured in situ value of injured natural resources. The reduction of in the situ use value is measured by the change in the value of harvesting or enjoying services of the injured natural resource, minus the change in the cost of harvesting or viewing the resource or visiting the area concerned. Damages are measured over the period of the resource recovery with all damages converted to the present value using a real discount rate of ten percent. The submodel does not include private losses, like reduced profits suffered by fish processors or coastal tourism hotel operators following a spill.

It is contended that CERCLA - as an unintended side-effect - provides effective market incentives for controlling pollution. For appraisal the behavioural responses of the polluting parties and the scope of the incidents have to be considered. Although there is only limited empirical assessment of a firm's behaviour under strict environmental regulation, it is suggested that the NRDA liability provision can create incentives for avoiding pollution. Here Becker's (1968) crime-and-punishment paradigm (detection, conviction, penalty, cost of punishment) may prove useful in the assessment of the behavioural response of economic agents engaged in prohibited activity. No empirical data, however, is available at present as to the effectiveness of CERCLA since its introduction in 1987.

B3.2.4 Future Developments

As far as transferable rights are concerned it should be stressed that such a system does not readily apply to sewerage and sewage disposal because important resource-use costs are involved with the sewerage and sewage disposal.

Boland (1986) describes a system of point source trading, applied with respect to discharges into the Fox River, Wisconsin, United States. The example will not be presented because this tradeable discharge permit system is only one element of the water pollution control measures aimed at cleaning the Fox River. It is therefore questionable whether general lessons can be learned from the Fox River experience.

B3.2.4.1 Example of Marketable Pollution Permits to Point/Non-Point Sources

Marketable pollution permits are applicable to combinations of point and non-point sources. Point sources could obtain additional rights by reducing the pollution burden from non-point sources, for example by financing best available agricultural practices.

A good example of point and non-point source marketable pollution permits system is the Dillon Reservoir in Colorado (USA). This water supply reservoir suffered from eutrophication. Four municipal waste water treatment plants were discharging their effluent after having made considerable reductions to phosphorus loadings. Non-point sources, mainly recreational and agricultural activities, were discharging their effluents into Dillon Reservoir as well. As a consequence, further reductions of phosphorous discharges were needed. Since non-point sources were difficult to control additional purification would have been required of municipal waste water treatment plants. Such efforts would have considerably raised costs of the waste water treatment plants.

In 1984 stricter standards were set for point source phosphorus discharges while higher amounts of phosphorus discharges were allowed when discharges from non-point sources were cut by amounts of twice the additional point source discharge (a 2:1 trading ratio). In the process of granting permits, non-point discharge decreases are not monitored, but the capacity of control equipment is estimated.

"Ex ante studies of the cost-effectiveness of a point/non-point tradeable discharge system concerned with phosphorus indicate that in the Dillon Reservoir area, cost savings of 50 per cent can be achieved compared to strict regulations. Although no actual experience is available, the system is expected to work well. Forced by regulations, water treatment plant operators can decide by themselves whether they will increase purification of their own discharges, say from 95 to 97.5 per cent, or install new equipment to control non-point discharges, say from 0 to 60 per cent, the latter being less costly, as the ex ante studies have indicated." (OECD, 1989a).

B3.3 CHARGES VERSUS PERMITS

Of the two preferred externality internalisation instruments we have introduced, which should be selected? Baumol and Oates (1979) present a strong argument in favour of consents, which runs along the following lines.

In principle, effluent charges and the trade of pollution consents lead to the same outcome: with effluent charges the regulatory agency raises the tax until the target level of emissions is achieved, while under the pollution consent scheme, it offers for trade, emission rights equal in total to the target amount. Moreover, the pollution consent approach has essentially all the advantages of effluent charges. It is dependable, because it is relatively automatic and routine, involving a regular monitoring of effluents to be checked against the polluter's registered number of emission rights. It is permanent: so long as it is not explicitly repealed, discharges will continue to be illegal without a permit. It is equitable in the sense that it requires polluters to pay for their waste emissions. It offers an incentive for maximal cleanup effort: every reduction in emissions reduces the number of consents the polluter has to buy. It involves a minimum of interference with private decisions. Finally, and perhaps most important, this technique has the same cost-saving advantages as the tax approach: firms for which pollution abatement is relatively inexpensive will find it cheaper to install abatement equipment than purchase pollution consents.

Some advantages of pollution consents over charges are:

Pollution consents are invulnerable to inflation. Since the quantity of pollution permits is fixed, their price will rise automatically in an inflationary period. As the value of money falls, polluters will bid more for each permit, so that no legislative action will be needed to produce a readjustment in the price of waste emissions.

As population and industrial activity increase, the demand for pollution consents will rise. If the number of permits offered for sale remains the same, all that will happen is that their price will be bid up; potential polluters will either have to outbid current permit holders or curb their emissions. The point here is simply that emissions cannot increase without an expansion in the number of consents provided by the environmental authority. Unlike the tax approach this scheme puts the burden of initiative on the polluters, rather than on public officials.

If one geographic area is more vulnerable to pollution than another, this can be dealt with simply by selling a smaller number of pollution consents (non-transferable between areas) for the first area than for the second. The sale of pollution permits may also cope better than taxes with temporal variations in the damage costs of pollution. For example, if emission into a river is particularly dangerous in the summer when the water is low, one can require the purchase of a special summer certificate for emissions; the environmental authority would issue a smaller number of summer permits. Since political pressures generally make for uniformity of treatment of different constituencies, it may not be realistic to expect the political process to yield significant differences in the abundance with which pollution rights are issued, even where there are good reasons for such differences. But, at least in principle, such variations are feasible.

The provision of a fixed number of pollution permits minimises uncertainty about the resulting level of emissions, at least so long as there is no significant quantity of illegal pollution. By contrast, the use of fees involves some uncertainty as to the outcome, because the public authority may not be able to predict with great accuracy the polluters' response to specific fees. Pollution consents eliminate this source of uncertainty by setting the level of emissions directly.

In one sense, the pollution consent approach may prove a less radical departure from existing programmes than a system of fees. The current direct control strategy embodied in present legislation regulates waste emissions largely through the issue of permits to individual polluters. The pollution permit technique could simply amount to making such permits transferable at a market determined price.

B3.4 ENFORCEMENT INCENTIVES

Enforcement incentives operate in conjunction with direct regulations and are economic instruments designed to ensure compliance with regulations. Two kinds of incentive were identified in the literature: non-compliance fees and performance bonds.

Non-compliance fees are levied against firms whose emissions exceed standards on a scale that is related to the profits reaped through non-compliance. They differ from fines in that their collection is not contingent on litigation. Consequently, they are more akin to effluent charges which become payable at the emission level of a standard.

Performance bonds are analogous to deposits. Polluting firms pay regulatory authorities prior to undertaking the polluting activity. If compliance with pollution standards is satisfactory, these funds are returned. The basis for return is usually not effluent emissions themselves but rather the installation of abatement equipment.

We briefly discuss the application of such instruments in a few countries.

AUSTRALIA

In Australia, non-compliance fees are a general provision of most environmental protection legislation. Fees, however, are not proportional to the potential economic value of non-compliance.

Performance bonds are extensively used to regulate the mining industry. Bonds are repaid upon satisfactory rehabilitation of the landscape following mining.

NORWAY

Non-compliance fees are levied against firms in violation of the Pollution Act. All types of pollution are covered by the fees. Prior to 1987 these fees were clearly too low and violations of the standards were increasing. In 1987 the fees were raised to more accurately reflect the benefit accruing to firms through the polluting activity. It is still too early to evaluate the impact of this measure.

SWEDEN

In terms of the Maritime Water Pollution Act, Swedish authorities may impose a non-compliance fee on ships discharging oil into the water. Charges are generally small although provision exists for charges up to $80 000. Since monitoring at sea is difficult, charges to data have mostly been applied to harboured vessels. Nevertheless, authorities report a decline in oil discharges and the measure is generally considered satisfactory.

United States

Under the so called "superfund" legislation in the US, polluters can be held liable for the cleanup cost of any environmental damage caused by hazardous waste. Thus, non-compliance fees are theoretically set at a cost equal to the damage caused and are thus similar to the liability instrument discussed above. Actual cases have resulted in very large costs to polluters and the instrument is felt to be both effective and efficient since firms are induced to review their waste management practices.

B3.5 Subsidies

One fiscal measure which we have not mentioned in our discussion of externality internalisation is the use of subsidies.

Subsidy proponents envisage a system in the developed world where firms are encouraged to abate pollution by receiving state subsidies for the installation of pollution control equipment. The arguments, in terms of the desiderata we have presented, against such a scheme are numerous and we do not present them exhaustively here. Suffice to say that subsidies, when intended as a pure incentive, are neither perceived to result in economic efficiency (they may even promote inefficiency) or adhere even to Standard PPP. It is, however, conceivable to consider subsidies in some cases, especially in developing countries. Consider, for instance, a situation where a polluting factory, providing vital employment in a given region, is truly unable to abate pollution to the level of a required standard. Forcing a given level of an environmental standard on the factory may result in costs that may be too high to justify and may result in loss of employment opportunities. In this case, a subsidy could be considered, not as an incentive measure, but to efficiently distribute pollution levels to achieve optimal pollution in a given area. In a similar context, use of subsidies can also be conceived of in rural areas, in order to support pollution abatement schemes that are too costly for a given community to implement. For further discussion of subsidies, especially for non-point-source pollution control, refer to section B4.3.3.2. of this appendix.

B3.6 CONCLUSIONS

International Lessons

Experience with economic instruments reveals a multi-dimensional concern that is not present in the purely theoretical treatment. While the goal of economic theory is unequivocally economic efficiency, reality dictates that policy makers be concerned with a wider political agenda. In the real world households and firms protest the imposition of new costs and taxes on their activities, central and local governments have to balance their budgets, social and economic programmes are competing with environmental programmes for resources, environmental damage costs are often not known, accurate monitoring of emissions is practically impossible and the list goes on.

Nevertheless, the international experience in environmental policy shows a definite trend towards the adoption of a new approach in which economic instruments occupy a central role. Two overriding priorities appear to have characterised this trend. Firstly, there is the recognition that industrialisation and economic growth have had a significant and deleterious impact on the environment. Accordingly, the first priority has been to ensure effective policy to safeguard human health and the overall quality of life through environmental protection. As awareness has increased, and standards have tightened, it has become apparent that such action is costly and creates a tension with the wider goal of economic growth. Thus, the second and more recent goal of environmental policy has been cost effectiveness. It is at this point that economic policy instruments have become more popular.

In this light we can interpret much of what is taking place in the environmental policy of developed nations. Whatever changes have come about, standards remain the mainstay of environmental policy and it is hard to see any shift from this basic position in the near future. Given the standards, the question has become: what is the most cost effective way to meet these goals? The answer to this question varies by industry and by resource.

In some cases, notably water treatment, the most effective approach appears to be public treatment facilities. These allow economies of scale in abatement processes to be reaped. Viewed in this way, it is natural that a charge (tax) and subsidy policy should come into being as indeed has happened in a number of countries.

In other cases significant cost savings can be realised by giving firms themselves the flexibility to decide how overall standards can be met. The emissions trading and liability insurance instruments appear to hold great promise for creating this flexibility and the large cost reductions that go with it.

The main point is that economic policies can be and are effective in the task of meeting environmental standards in a cost effective fashion. It is clear though, that no one instrument is a panacea. Different instruments will be appropriate in different contexts.

Besides being cost effective and perhaps because of it, our survey has also shown evidence to suggest that economic instruments can be effective in achieving compliance with the standards. Charges in the Netherlands and Germany have had a strong incentive impact and have resulted in real environmental improvements. Deposit systems seem to yield similar results. At the same time the evidence also suggests that policies may not be effective if they are poorly implemented. The failure of the Fox River trading scheme due to over regulation of the trading practices and the under incentive effect of many European charge schemes due to low charges are reminders of these dangers.

Finally, it is encouraging to note that economic instruments have gained acceptance as a means of environmental regulation in developed countries. In many cases this acceptance has not come easily and has been resisted by industry. In the end however, there are usually palatable ways of introducing such schemes. Putting a charge scheme in place may mean starting with low charges to be increased later, as the Germans have done. Or, a tradable permit scheme may start with generous initial allowances which can be later reduced by public purchase of pollution rights, thus creating an environmental industry.

B4. POLLUTION CONTROL METHODS FOR NON-POINT-SOURCE POLLUTION AND SEDIMENTATION

As the name suggests, non-point pollution (NPP) originates from diffuse sources and cannot generally be traced back to a single source such as an effluent outlet pipe or a smokestack. This physical characteristic leads to a number of problems in the formulation and implementation of feasible control policies. Clearly, the economic incentive policies outlined in previous sections are inadequate because they implicitly assume that the source and quantity of discharge is known.

This section will attempt to look at the difficult problem of controlling non-point sources of pollution, reviewing most of the known policy approaches instead of restricting the discussion purely to the so-called economic approaches. Before launching into a discussion of policy measures, brief descriptions of the basic characteristics and sources of non-point pollution are warranted.

B4.1 SOURCES OF NON-POINT POLLUTION

Five major sources of non-point pollution are commonly identified in the literature: sediment, plant nutrients, pesticides, urban runoff and mining activities. In addition, leaching from certain hazardous sites such as landfills, farm building and industrial premises also contributes to the overall impact of non-point pollution.

Sediment is without doubt the largest non-point pollutant when considered by volume. While one may not immediately think of sediment as a pollutant in the conventional sense, it should be remembered that the loading of waterways with sediment imposes measurable economic costs and it is therefore appropriate that it be considered in the same manner as any other pollutant.

Plant nutrients, notably phosphorus and nitrogen, originate in fertilizers and lead to the eutrophication of lakes and streams as they migrate through the soil into waterways.

Modern *pesticides* are increasingly soluble and toxic and are being transported larger distances off farmland into surface and groundwater supplies.

Urban stormwater *run-off* is an increasingly important and toxic source of non-point pollution. Sewage, oils, heavy metals and numerous other toxic substances which accumulate in urban areas are washed into the drainage system. In South Africa, the prevalence of informal settlements contributes significantly to this type of water contamination.

Mining activities leave behind large quantities of processed material on dumps which frequently contain toxic substances that are leached out into precipitation run-offs.

B4.2 Characteristics of Non-point Pollution

Three characteristics of non-point pollution distinguish it as a separate policy problem.

1. In all the major instances of non-point pollution there is a great deal of uncertainty surrounding the environmental effects of the polluting activity. In most cases the relationship between the discharge (e.g. amount of pesticide applied) and the pollution effect (toxic effects in a river) are far from clear. A variety of environmental factors such as varying soil types contrive to make this uncertainty even more acute

2. The nature of most non-point pollution is such that individual polluters cannot be monitored and only the final effects are observable. This, together with the uncertainty means that it is generally impossible to attribute final damage to polluting individuals.

3. The monitoring problem is further exacerbated by the fact that non-point pollution is usually generated intermittently, e.g. following a storm, rather than continuously.

It is, therefore, a principal characteristic of non-point pollution that neither the polluter nor the regulator is fully aware of the true extent of the pollution discharge.

B4.3 Policy Options for Non-point Pollution Control

If the preceding sections have shown anything, it is that non-point pollution is both widespread and almost impossible to monitor due to inherent monitoring difficulties and the high degree of uncertainty surrounding it. Despite the almost overwhelming challenge this creates for an environmental authority there is hope that the problem can be approached. Chesters and Schierow (1985) make the encouraging observation that "technical solutions are available for virtually every known non-point source pollutant". At the Chief Directorate: Sea Fisheries of the Department of Environment Affairs in Cape Town for instance, oil crossmatching techniques are used to match samples from (fresh, unweathered) oil with samples from a suspected 'culprit'. In most cases of oil pollution in an aquatic environment, the source can actually be determined. The techniques used can theoretically also be used for spills on land. Given the basic premise that technical solutions to trace non-point source pollutants are available, what remains is to devise policies which encourage polluters to adopt these solutions in a cost effective manner. To date three broad classes of policy measure have been advanced[3].

B4.3.1 Voluntarism

Such policies eschew the use of direct incentives and penalties and instead aim to encourage polluters to voluntarily engage in abatement activities. Such policies can be highly effective (Harrington et al. 1985) when:

- Polluters do not incur large costs through such activities;
- When they might potentially incur large cost through not engaging in such activities, i.e. in the case where a firm's sales might be harmed through a poor environmental image;
- There is general agreement that the goal is a worthy one and that the action sought will achieve the objective,
- non-compliance is easily observable so that social pressures for compliance are created,
- There is a belief that failure to tackle the problem will result in mandatory government action.

3 The following policy discussion is largely ba non-posed on the article by Harrington et al (1985). This appears to be the leading work onint pollution policy and its lack of rigour and depth point to the general weakness of research in this area.

Generally, voluntary participation in abatement activities is achieved through technical assistance, education or moral suasion.

Technical Assistance

Voluntary policies may provide polluters with technical assistance on applying pollution abating practices. Such services might be applied in a variety of areas from providing basic training in soil management practises to instruction on the construction of pit latrines.

Education

Sometimes education may persuade polluters that polluting activity is not in their own long term best interests. Farmers generating sediment pollution, whose long term interest is the preservation of topsoil, are good examples of such polluters. Umgeni water has recently instituted another interesting example called Project WATER in which informal settlement dwellers are provided with the means to test their water supplies and through the ensuing awareness take basic steps to safeguard them.

Moral Suasion

Polluters are made aware of the detrimental effects of their actions and are encouraged to take appropriate steps to rectify the problem.

B4.3.2 Command and Control

Where voluntary programs are ineffective direct regulation of non-point emissions must be considered. One of the methods open to decision-makers when tackling pollution issues is the use of so-called command-and-control policies (or means of distributing control responsibility among points of discharge). Such policies, reflected in regulatory legislation, aim at the reduction or even elimination of the external costs associated with pollution discharges through the setting of certain specific targets or standards. However, it is found, that the amount of information needed to set appropriate standards for each plant and each source of emission is not only too high to be coped with by the decision-maker, but also typically not available when allocations are made. Plant managers tend to be in the opposite position, knowing fairly exactly the control mechanisms needed for cost-effective emission reduction and cost savings possible. The big dilemma with the command-and-control approach is generally perceived to be the discrepancy between the capability of the polluting agent and the responsibility of the controlling agent. Given the divergence of information available and the difference in incentives to accept control responsibility (the polluting agent having virtually none) the command-and-control approach by itself has little chance of becoming cost-effective. It can, however, be used in conjunction with other policies and tends to be useful when dealing with non-point-source pollution.

If they are used, these "command and control" measures are usually implemented through a series of regulations, often accompanied by a permitting process and enforcement monitoring and fines. Legislated regulations typically specify either performance or design standards that form the two basic classes of command and control regulations.

Design Standards

Design standards specify practices or designs that are to be employed by potential non-point polluters. For example, building permits might specify land management practices to minimise soil erosion during construction or farmers might be obliged to employ contour ploughing.

It is generally recognised that the enforcement of design standards is a difficult and potentially costly process. Furthermore, there seems to be potential for the costs of implementing the standards to fall heavily on a few parties. In some industries, such as internationally competitive agriculture, this could adversely affect competitiveness and lead to the demise of producers as they are unable to pass along the increased production costs. Consequently, it is likely that the introduction of many regulatory policies, effective as they may be, will not be politically viable.

Performance Standards

Performance standards regulate the actual amount of pollution discharge and leave the management of emissions to the polluter. As such, these standards are inherently more cost efficient than design standards. Unfortunately, however, they are generally not feasible for non-point pollution because of the difficulties in measuring emissions from individual polluters.

One interesting possibility is to use agreed rules of thumb such as the universal soil loss equation (USLE)[4] which specifies the relationship between soil loss and six easily measured variables encompassing: climate, topography, cropping systems

4 This widely recognised formula is documented in Wischmeier and Smith (1978).

and management practices. Should such a rule be used, regulators could then evaluate an individual's performance using easily measurable proxy variables in lieu of direct measurement.

B4.3.3 Economic Incentives

The final class of measures, and from the economist's standpoint the most appealing, is economic incentives. These may assume various forms and are discussed below.

B4.3.3.1 Taxes and Fees

As in point source pollution the general idea here is to increase the cost of generating pollution and thereby pre-empt an emission's reduction.

The overt, direct use of fees for controlling non-point pollution is unknown in the United States or anywhere else as far as can be established. The reason for this is probably related to measurement difficulties and the associated assignment of fees. Nevertheless several studies (see for example Miranoski et al. 1982) indicate the existence of a significant price response relationship.

A more practical approach is the indirect application of fees. Under such a system, charges would be levied on products or processes that lead to non-point pollution. For example, a tax would be levied on fertiliser rather than on the (almost impossible to measure) additional nutrients that enter the water system as a result of its application. If charges are appropriately set, fertiliser use will be reduced with a concomitant reduction in non-point source pollution.

Another possible use of fees would curb erosion on construction or new township sites. A land disturbance fee, collected on application for a building permit, based on the amount of land disturbed would discourage contractors from disturbing more land than is necessary for the immediate work at hand. In this way, building could progress incrementally without leaving vast areas exposed to erosion for the duration of a project.

B4.3.3.2 Subsidies

Where taxes represent the stick approach to abatement, subsidies offer the promise of a carrot. Payments in the form of direct transfers, guaranteed prices, tax exemptions, low interest loans, etc. are used to induce compliance with environmental goals.

Subsidies themselves can be paid in response to actual pollution reduction, the adoption of certain desirable practices or to cover the cost of an abatement investment. They have the advantage that they are more politically acceptable than other measures and consequently have been widely used for decades, particularly in agriculture.

American experience with soil conservation subsidies illustrates the care that must be taken in their application. Batie (1983) estimates that more than half of all soil conservation subsidies in the US have been applied to land where erosion is not a significant problem. This situation arose through a political process which emphasised equity for individual producers. Clearly this is an inefficient use of public funds.

The usual range of problems associated with subsidies applies as much for non-point pollution as for any other area. Payments are made to people for actions they would have taken anyway, the production input mix is distorted, long term investments are made on the basis of a subsidy making it difficult or impossible to revoke them and numerous unintended and undesirable effects often result. Most importantly, there is the question of the cost of the subsidies and who pays them.

B4.3.3.3 *Tradable Permits and Bubbles*

A very innovative and potentially cost effective policy involves tying non-point and point pollution sources together in a so-called bubble. Bubbles are geographic areas in which total effluent loading from all sources is targeted at some level. Each polluter in the bubble is issued with a "tradable permit" specifying the amount of pollution loading to which they are entitled. Polluters who find it too expensive to abate their emissions may seek to purchase permits from other polluters who are able to abate more cheaply and then sell their additional permit rights to cover the costs. In this way, total pollution levels in the bubble are controlled at minimum overall cost to the local economy pollution.

Existing tradable permit systems, because of the measurement issues, normally only apply to point sources. Recently, however, it has been suggested that point source polluters[5] might benefit by having the opportunity to obtain their pollution offsets

5 See the previous section on international experience for a description of the Dillon reservoir scheme in Colorado for an illustration of the practical application of such a policy.

seeking to meet its pollution quota may either buy rights from another point source polluter or pay a non-point source polluter to undertake abatement activities. Because of the uncertainty involved in non-point emissions it is usual that the non-point pollution reduction exceeds the point increase by some fixed ratio, typically 2 to 1. The burden of proof in such a system would usually be on the point source who would be required to demonstrate that the proposed reductions in non-point emissions are adequate and effective before the regulating authority's consent is granted.

This bubble trading system is usually only effective when point and non-point sources coexist in a geographic region in which total pollution loads can be measured. Such circumstances may exist whenever agricultural and industrial enterprises are in proximity to one another or when a number of sources contribute to the pollution of a lake, river system or estuary.

The system we have outlined above places the bulk of the financial burden on point source polluters and as such may be perceived as inequitable. Further refinement of the system, based on rules like the universal soil loss equation, might redress this balance by requiring some abatement action from non-point polluters thus inducing them to share in the cost of the bubble's pollution reduction.

A number of implementation issues generally need to be resolved before a permit system can become viable policy[6].

a) Target loadings for the bubble need to be established. Economic theory suggests that this level should be some "optimal pollution level" based on an economic cost benefit evaluation. Practically, it appears more desirable to make use of scientific and social studies to establish prudent levels.
b) It must be possible for an agency to monitor point discharges and overall pollution loading in the bubble.
c) Institutions to facilitate trading must be created.
d) Considerable technical information is required to establish the boundaries of a bubble.
e) There are significant ethical and political problems in deciding on the initial allocation of permits to polluting agents.

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6 These issues are basically the same for any permit system irrespective of whether non-point sources are included. Reader who are interested in these issues should refer to the section on tradable permits.

e) There are significant ethical and political problems in deciding on the initial allocation of permits to polluting agents.

B4.3.4 *Adjusting Government Policies*

Many existing government policies have indirect or unintended impacts on non-point source pollution. Principal among these is aid to the agricultural sector in the form of price supports, soil banking financial subsidies, etc. Other policies include tax depreciation policies, zoning ordinances and governmental land and water programs.

For example, excessive price supports or unduly cheap water may result in the cultivation of marginal land, ultimately leading to soil erosion. Overpopulation of subsistence areas is another source of soil erosion which has indirectly been affected by national policies.

B4.3.5 *Cross Compliance*

Cross compliance is a hybrid policy which appears to be quite promising in the regulation of non-point emissions. The idea is to link access to other government programs such as farm subsidies with reduced non-point pollution. The required reduction measures will usually assume the form of particular mandated practices.

Care must be taken in such schemes to ensure that the compliance costs are not so high as to dissuade polluters from participating in an otherwise beneficial program. It is also not clear, that cross compliance results in the most cost effective abatement of non-point pollution.

B4.4 CONCLUSION

In the sections above we have taken a brief look at the problem of non-point pollution and the some of the policies that have been advanced to address these emissions.

It is immediately clear that very little legislation is actually in place to address this form of pollution. In a large part this is due to the complexity of the overall problem and the lack of real theoretical foundations to policy in the face of the monitoring problem.

Nevertheless, there are both technical solutions and policies to combat the problem. No single policy is a panacea to the problem and clearly all potential policies are limited by particular circumstances. Consequently, it seems logical that an effective non-point pollution strategy will in it essence be eclectic, drawing on different policy prescriptions to combat the various geographic, socio-economic and physical circumstances. Clearly more detailed work will have to be done to arrive at a policy suited to the South African context.

B4.5 POINT-NON-POINT CONTROL TRADE-OFF ISSUES: AN EXAMPLE FROM DU PAGE COUNTY, ILLINOIS[7].

The preceding sections have shown that relative costs and water quality effects of controlling point- and non-point source pollution are important issues in environmental policies around the world.

One of the major targets for point-source pollution control by environmental authorities in the United States have been municipal wastewater treatment plants. Concern, however, has developed as to the efficiency of additional large financial investments in more advanced treatment processes of such point-source pollution: effluent standards for municipal treatment plants are found to be excessively strict and only small improvements in water quality are expected to ensue from further investment. As a result non-point sources of pollution have become the prime factors in deteriorating water quality. Following from this there appears to be increasing need for an improved investment strategy for treatment of non-point source pollution. Effective management of such pollution comprises of run-off control and pollutant concentrations reduction. Requirements are systems for water storage and treatment, and land management practices which are appropriate for a given community.

The study presented here compares the costs of improving water quality through control of point- and non-point-sources of water pollutants and considers pollutant loads of point- and non-point origin into the DuPage River basin, Illinois, US. Two major sources of diffuse pollution loads were identified: run-off discharged through storm sewers and overland run-off, especially from grassland areas. It was found that both pollutant source associated with run-off could be drastically reduced.

7 This section is essentially based on: Macal, Charles M, Broomfield, Barbara J., Point Versus Nonpoint Pollution Control Strategies, in: George S. Tolley, Dan Yaron, Glenn Blomquist, Environmental Policy, Water Quality, Volume III of Environmental Policy, A Five-Volume Series, Ballinger Publishing House, Cambridge, Mass., 1983, pp.163-182.)

Three strategies, recognised as having the greatest practical potential for implementation to reduce the pollutant load are briefly described here:

1. Vacuum sweeping of streets and parking lots. This was expected to reduce the BOD (Biological Oxygen Demand) in run-off for existing urban areas by 25% and by 50% for newly urbanised areas. Additionally, gradual implementation of BMPs (Best Management Practices) for agriculture was envisaged.

2. Control options for over-land run-off. The reduction of shock loading impact through the construction of a number of retention basins along the streams.

3. The use of porous pavements which are expected to be effective in regulating run-off from streets and parking lots.

To determine water quality effects of various treatment plant standards and non-point source controls the authors made use of a dynamic water-quality/hydrology simulation model and went on to assess the cost-effectiveness of point- and non-point-source strategies considering costs and water quality data. Findings show, that controlling non-point-source pollution in this given case yields more cost-effective results and improves water quality beyond the scope of wastewater treatment plants.

Associated problems with this analysis, however, are that representative cost and performance data were used and no site-specific analyses were carried out. Furthermore, the costs associated with such analyses have to be included in the appraisal of the cost-effectiveness. Furthermore institutional and social impediments have to be overcome before such strategies can be implemented.

APPENDIX C

Economic Instruments for General Evaluation Purposes

CONTENTS

C1. COST-BENEFIT ANALYSIS

C1.1 THE CURRENT STATUS OF CBA

Cost-benefit analysis (CBA)[1] has now been practised internationally for almost half a century, owing its origin as an analytical technique to the USA's Flood Control Act of 1936 l(Pearce, 1983, 14).

1 CBA is a procedure for measuring gains and losses to individuals, using money as the measuring rod of those gains and losses, and aggregating the money valuations of the gains and losses of individuals and expressing them as net social gains and losses. A related technique, cost-effectiveness analysis (CEA), measures benefits in some appropriate physical units, or simply states them as a policy objective, while costs are measured in monetary units.(Pearce,1983,3&15).

It has been used mainly in the context of public sector project and policy appraisal, although in principle there is nothing to prevent it being used to analyse private sector investment decisions as well, in which case it would indicate the extent of divergence between private and social economic efficiency. The reasons for this are related to the discussion of pricing that was presented in earlier chapters of this document, and may be said to be either that those commodities for which a straightforward pricing solution is impossible or politically unsatisfactory will generally be those that find their way into the public sector, or that the public sector is supposed to be more concerned with the broader general welfare of the population[2].

Throughout the period of its use, there has been a groundswell of dissatisfaction with CBA's approach of expressing all the benefits and costs of some action in monetary terms, especially when there are no markets for the benefits and costs in question. Indeed, as will be discussed below, it can be argued that there are real disadvantages in broadening CBA into an all-encompassing technique, and that it is unnecessary to do so.

Leaving this debate aside for the moment, however, it is fair to say that, notwithstanding this dissatisfaction and the oft-stated objections to which it has given rise (for a detailed discussion of these see Campen, 1986), CBA has gone from strength to strength both in the frequency of its application and the range of problems to which it has been addressed (discussions of these can be found in Pearce,1983, and Campen,1986). Pearce,1983,21 explains the success of CBA ("a relentless advance by CBA and CEA into the appraisal of government expenditures") as follows:

2 There is something vague and unsatisfactory about this distinction. A commodity will often be allocated through the public rather than the private sector for reasons of historical accident rather than any inherent logic rooted in its value to society. Consequently, the application of CBA as opposed to pricing may be anomalous. For example, road provision - which in developed societies with mature infrastructure networks at least would seem to require pricing - will usually be subjected to CBA because it is a public sector function, while the provision of basic needs such as food and housing will be left to the private sector and hence to arguably unsuitable allocation via pricing, albeit with substantial government intervention under the banner of "development policy". It may be wondered whether the use of pricing rules in allocating roads, and welfare-based CBA in allocating commodities that meet basic needs, would not lead to better results in both cases. Such an approach would be consistent with that adopted in this document, where it is the dominance of merit or commercial characteristics that determines whether pricing or CBA should be used to allocate water amongst competing uses.

> Arguably, the explanation is very simple. Those who practised CBA had a real-world task to attend to. Someone had to decide on the priorities within any sub-budget of government expenditure. The niceties of academic interchange in the learned journals did little to aid those who had these tasks. Instead, it seemed that not only did CBA offer a technique for aiding the evaluative process, albeit subject to many caveats, it actually offered the *only* reasoned technique. CBA also had a fundamental attraction of reducing a complex problem to something less complex and more manageable.

Notable events in the history of CBA have been the publication of several manuals: by the OECD (Little & Mirrlees,1974), UNIDO (Marglin et al,1972) and the World Bank (Squire & van der Tak,1975). These provided standardised guidelines for the application of the technique, and were intended especially for use in less developed countries.

Notable, too, in the present context has been the prominence that problems related to water management have assumed in extending the boundaries of knowledge in CBA. Seminal works have included the "Green Book" produced by the US Federal Inter-Agency River Basin Committee in 1950; Eckstein's *Water Resource Development* , Krutilla & Eckstein's *Multiple Purpose River Development* , and McKean's *Efficiency in Government through Systems Analysis: With Emphasis on Water Resources Development*, which were all published in 1958; and the *Design of Water Resource Systems* (Maass,1962) produced by the Harvard Water Resource Program.

For much of CBA's history, however, some of the strongest opposition to its use came from early advocates of the environmental movement. To these people, in particular, the expression of all benefits and costs - including environmental ones - in monetary values was anathema. With the growth in importance of environmental considerations during the seventies, this opposition might have been a deathblow to a less robust technique, and indeed its popularity declined for a time. Once more, however, CBA withstood the challenge. A spate of work saw new methods being devised to value features of the natural environment. The following extract from a publication by the US Environmental Protection Agency (*EPA's Use of Benefit-Cost Analysis*, Washington DC, 1987, quoted in Pearce, Markandya & Barbier,1989,123) gives an indication of how deeply it is now entrenched in decision-making concerning the environment:

> Among the many ways that benefit-cost analyses have influenced the development of regulations at EPA are the following:

(i) Guiding the regulation's development,

(ii) Adding new alternatives,

(iii) Eliminating non-cost-effective alternatives,

(iv) Adjusting alternatives to account for differences between industries and segments

(v) Supporting decisions.

> At times benefit-cost analysis has led to more efficient regulations by showing how more stringent alternatives would bring about a greater reduction in pollution without a commensurate increase in costs. In two instances...this led to the adoption of regulations that were more stringent than originally contemplated. At other times the analysis showed that the costs of more stringent regulations would be disproportional to the expected benefits. In three instances...this led to the selection of less stringent regulatory alternatives that resulted in reduced regulatory burdens without significant reductions in environmental improvement.
>
> While these improvements cannot be attributed solely to benefit-cost analysis, it is fair to say that the analyses played major roles in bringing about the regulatory improvements...
>
> The contributions of the benefit-cost analyses prepared by EPA go beyond individual regulations, however. In addition to improving individual environmental regulations, benefit-cost analyses also have increased awareness of the environmental results of EPA's regulations, provided a framework for comparing regulations both within a single medium and across media, identified cross-media effects, and improved analytic techniques.

From this discussion it is clear that in the environmental context CBA has come a long way over the past two decades or so. The situation has changed diametrically from the one prevailing in the early seventies, in which the weighing of the benefits of proposed environmental standards against their costs was explicitly or implicitly shunned. As Cropper & Oates, 1990, 44&92-93, point out, this change is in large part a reflection of the evolution that has occurred in environmental management itself, leading to a perception of the sizeable and rising costs that are imposed by protective measures. The need for systematic CBAs is likely to continue to grow:

> The role for economic analysis in environmental policy-making is far more important now than in the earlier years of the "environmental revolution". When we set out initially to attack our major pollution (and resource) problems, there were available a wide array of fairly direct and inexpensive measures for pollution control. We were, in short,

> operating on relatively low and flat segments of marginal abatement cost (MAC) curves. Things have changed though. As nearly all the (empirical) cost studies reveal, marginal abatement cost functions have the typical textbook shape. They are low and fairly flat over some range and then begin to rise, often quite rapidly. Both the first and second derivatives of these abatement cost functions are positive - and rapidly increasing marginal abatement costs often set in with a vengeance.
>
> We now find ourselves operating, in most instances (in the US), along these rapidly rising portions of MAC functions so that decisions to cut pollution (or conserve resources) yet further are becoming much more costly. In such a setting, it is crucial that we have a clear sense of the relative benefits and costs of alternative measures. It will be quite easy, for example, to enact new, more stringent regulations that impose huge costs on society and yield very little in the way of benefits...to the citizenry...

Turning from this review of the current status of CBA in the more developed countries to consider its status in South Africa, one finds that this country has lagged behind in its application of this technique. As a general observation it can be stated that CBAs have to date been employed only sporadically as decision-making aids, and those that have been carried out have often been relatively unsophisticated. Times are changing, however. Under the new budgetary discipline that is being imposed as a result of the straitened economic circumstances in which the country finds itself, CBAs are being demanded more and more frequently. When one adds to this the impetus that the technique, judging by what has been said above, is likely to acquire from the acceleration of the environmental debate in South Africa, it seems fair to suggest that CBAs will be conducted with much greater frequency in future. Furthermore, the significant benefits that CBA's offer have not yet been tapped to any real extent.

In the light of CBA's traditional association with problems of water management, it therefore seems reasonable to suppose, firstly, that the technique has much to offer in this field and, secondly, that it has not been fully exploited for this purpose in this country.

In section C.2 below, attention turns to an investigation of the validity of this supposition. First, however, it is necessary to investigate in more detail the limitations that CBA has as an analytical technique.

C1.2 LIMITS TO THE CBA METHODOLOGY[3]

Having argued in the preceding section that it is likely that CBA will be able to make a greater contribution to water management decision making in future than it has done in the past, it is necessary to qualify this conclusion carefully. This is because tendencies are emerging in the use of CBA that the authors of this report view as dangerous, and they would not want to see them introduced into the water management arena.

The CBA methodology can be interpreted in ways that range across a wide spectrum. At the two ends of this are found what can be referred to as "narrow" and "broad" CBA, supported respectively by the "conventional" and "decision-making" schools of writing on the technique. The distinction between them lies in the role played by decision makers.

The "conventional" school maintains that CBA should be based only on objective (market) data and generally accepted principles of economic efficiency. Proponents of broad CBA, by contrast, hold that decision-maker's more subjective beliefs concerning, for example, weights of benefits, social and political factors, and the values of non-economic variables, should be included as part of CBA.

As was noted in section C1, CBA has to date been used in a fairly desultory fashion in South Africa. Its introduction on a wider scale is, however, being envisaged, and a manual containing a set of common operating practices has been prepared by the Central Economic Advisory Service (CEAS). This manual, though not very explicit on the matter, generally favours a broad approach to CBA, and this is the first danger to which attention must be drawn here, as will be done in the discussion below. Compounding the error, however, is the rationale of the CEAS manual itself, which is to provide a "recipe book" so that project analysts, trained in disciplines other than economics, can apply CBA at a decentralised level of decision-making by following rules laid down by central government.

It is the opinion of the authors of this report that if this combination of decentralisation, analysts who are non-economists and broad CBA is a recipe for anything, it is a recipe for disaster. To support this view, one may begin with what

3 The discussion in this section is based on Dockel, Mirrilees & Curtayne, 1990. The first two authors were also amongst the compilers of this present report.

Dockel, Mirrilees & Curtayne, 1990, 8-9, argue to be self-evident propositions concerning the use of CBA in South Africa.

> The strongest argument for the use of CBA is that for many purposes there is no better alternative. It is important to restrict the technique to those situations, however. It cannot be considered to provide an automatic allocative rule to substitute for the complexities of budgetary decision-making.
>
> CBA is at best a relatively "soft" technique, and it becomes softer as it is extended to encompass greater proportions of non-measurable costs and, especially, benefits.
>
> It also becomes softer as the degree of uncertainty about future economic states increases.
>
> Being a soft technique, it is open to accidental or deliberate exploitation such as the pseudo-objective justification of projects favoured for political ends.
>
> CBA is also a complicated technique to apply, and so should best be left to fully trained evaluators in cases where subjective judgements are necessary so that those judgements will be properly founded and readily defensible. This implies that wherever subjective judgement is required, the analysis should be done by a central organisation using internally agreed upon and consistent methods and protected from the pressures of local political and bureaucratic influences.
>
> While we find it laudable that organisations such as CEAS should seek to standardise CBA procedures, we believe that the degree to which this undertaking will permit the performance of decentralised CBAs will always be hampered by inherent limitations which should be acknowledged:
>
> - Not all possible situations can be treated
> - Conditions can change drastically within fairly short time frames in a dynamic economy
> - The attempt to specify socially optimal prices through the prescription of accounting (shadow) prices that purport to counter the distortions that occur on perfectly competitive markets as a result of taxation, tariffs, monopolies, unemployment and so on, is potentially dangerous, particularly in a dynamic situation. There are no a priori grounds for believing that these accounting prices are superior to those generated by an imperfect but nevertheless well-developed market economy

- On the output side where market prices do not exist (for example, values for human life), accounting prices can be notoriously inaccurate
- While the use of accounting prices may be appropriate, even unavoidable, in certain circumstances, the subtleties involved in their use are too considerable to be entrusted to non-expert practitioners by way of mechanistic prescription.

> Finally, the CBA technique is well suited to deal with efficiency considerations, despite its softness even in this more modest context. It does not, however, have any particular merits when equity, strategic or other social issues are to be considered and when, as a result, value judgements predominate in the decision-making process.

Against this background, the idea of broadening CBA as envisaged by CEAS does not fare well. Certainly if the technique is to be used at a decentralised level by non-economists the view that it can encapsulate all or most of the decision-making process overall is not appealing. As Dockel, Mirrilees & Curtayne, 1990, 4, argue -

> We do not believe that CBA is the best technique for dealing with the "softer" aspects of decision-making, especially because it is a single criterion method and these aspects invariably involve attempts to satisfy multiple goals. And we dot believe that the more difficult subjective aspects of decision-making become easier to deal with when they are forced into the confines of a supposedly objective vehicle and then delegated to the non-expert. To put this another way, we believe that CBA can provide one important input into the decision-making process, but not determine the outcome of that process. It should be treated as a decision-making tool, not a decision-making rule.

Strong support for these views can be found in the literature. Mishan,1988,209, for example, says -

> It cannot be too often stressed that cost-benefit analysis as traditionally practised is no more than a useful technique in the service of social decisions...
>
> Attempts to work more into the technique ... to endow it with greater self-sufficiency for policy purposes by recourse to distributional weights or national parameters formulated by reference to political decisions or, at any rate, by reference to non-economic considerations ... entail (many) disadvantages...

And Campen,1986,83, agrees that -

> CBA is merely one input into the outcome-determination process. The role of CBA is to provide better information to decision-makers, to

encourage systematic thought about alternatives, consequences, and values, and to increase the explicitness of the evaluative process and the accountability of those who actually make decisions. Decision-makers must still use their judgement to combine the economic efficiency results of CBA with the results of evaluations according to other relevant standards - whether these standards are concerned with ethics, jurisprudence, politics, distributional justice, or ecology. Quantitative information provided by CBA constitutes a highly useful input into the process of balancing, or trading-off, among different objectives - a necessary process whenever there is no dominant alternative that is best with respect to each individual standard.

The conclusions of this line of argument are twofold. Firstly, CBA is a good and valuable technique with which to determine, from a set of feasible decisions, the one that is most economically efficient. But it is not a sound basis on which to decide which decision is optimal from the viewpoint of non-economic criteria. For this latter purpose, other techniques (multiple criteria decision making or MCDM techniques) have been developed, and for a comprehensive analysis of a decision CBA should be treated as one input into these MCDM methods, and not as a substitute for them.

Thus, when CBA-type techniques for valuing the environment are presented below, they must be viewed as providing an exclusively *economic* value for the environment. This may need to be modified within the MCDM framework, if it is believed that the environment has other values in addition to the economic one that people place on it, but this is not something that economists can either express an opinion on or measure. The same argument holds good where values are described to the equity criterion, or to any of the other criteria that may be encapsulated by the MCDM procedure.

CBA is thus able to embody much the same set of criteria as MCDM. In doing so, however, it must express them in monetary values and so converts them to a single criterion: their economic worth. To the extent that it may be believed that the criteria have other dimensions than economic worth, it will be necessary to overrule the economic efficiency decision preference of CBA, by incorporating it as just one option among many in an MCDM analysis[4].

4 It should be pointed out in this regard that to argue that MCDM will capture additional values of, say, equity or the environment that elude a rigorous CBA is to suggest that people have desires that they are not prepared to back up with money. For example, they have a need for a feature of the environment to continue to exist, but are not prepared to pay for its preservation. It is difficult to find any logic in this argument, which assumes that certain needs, or wants, are somehow qualitatively different from others. Hence the purist economic view that "values" that cannot be incorporated into

CBA because they cannot be expressed in monetary terms are spurious and cannot be taken seriously in a world of scarce resources and competing uses for them. Be that as it may, however, it remains true to say that from a pragmatic point of view there are tasks of valuation that will be more convincingly dealt with, at least in the eyes of non-economists, by way of MCDM rather than CBA, and this must be recognised if the ultimate aim of the exercise is to reach the best decisions possible.

C1.3 Present and Potential Use of CBA In Water Management in SOUTH Africa

To date, DWA&F has used CBA for the evaluation of infrastructural developments; that is, new impoundment and irrigation schemes. The technique has been used in its narrow form, as described in the previous section, and in fact has been limited to the establishment of direct monetary costs and benefits. The decision rule employed has been that where these benefits are roughly sufficient to balance the costs, the proposed project has been approved on the assumption that indirect and non-monetary benefits would be sufficient to tip the scales.

This is a sound and pragmatic way of making use of CBA as a decision-making aid, and apparently has produced intuitively acceptable results. However, the decision-making milieu seems set to become more complex in future, as the availability of supply-side solutions to situations of water scarcity becomes increasingly constrained by either affordability or environmental limitations. This suggests that the decisions that will have to be taken in future will be of a different type. The question to be addressed here, then, is whether CBA shows promise in such less-traditional situations.

That this is indeed the case is demonstrated amply by the following list of six potential application areas, which is almost certainly not exhaustive.

Quality standards.

A form of CBA for environmental policy evaluation in the USA. Called Regulatory Impact Analysis (RIA), this has had some dramatic effects, and in several cases has led to savings to the US economy that have been estimated at hundreds of millions of dollars.

It seems clear that with the recent introduction of the Receiving Water Quality Standards (RWQS) system for the control of water quality in South Africa, CBA (or RIA) could make a valuable contribution by calculating an optimal level for the standards that are to be introduced. At present, the system does not appear to make provision for the rigorous calculation of the costs and benefits that would accrue to different parties in given catchment under alternative levels of standards; in other words, precisely that analysis which has produced such substantial savings in the USA is left out of the RWQS system in this country.

Catchment management.

CBA, or perhaps more correctly CEA, also seems to offer the scope for substantial improvements in the cost-effectiveness of certain catchments. For example, the Umgeni catchment is often said to have a "water quality problem". This rather vaguely defined issue upon closer investigation emerges as largely a problem of comparative costs. As a result of fragmented institutional arrangements in the region, the various parties responsible for water utilisation take a parochial view of cost minimisation: each attempts to minimise its own costs, irrespective of the consequences for other users. Observation by people involved in water management of the Umgeni suggests that the result, as one would expect, is the creation of excess cost burdens in the region as a whole, which are imposed in particular on downstream users.

An obvious response to this situation would be a region-wide cost-effectiveness analysis; indeed, it is surprising that this has never been undertaken, as the potential cost-savings that could be realised are clearly quite large compared to the costs of the study itself. The cost-effectiveness analysis, would document the main water management options which exist for the region, along with their attendant costs, and then seek to develop a least-cost strategy, possibly accompanied by transfers of revenues between local authorities.

Intra-catchment allocation.

Earlier sections of this document have argued forcibly that the scarcity of water that is beginning to be experienced in many South African catchments requires the introduction of effective pricing procedures. Over the longer term, this is believed to be the only feasible approach. Past practices have produced major inefficiencies in allocation, however, and these have their own substantial inertia. Even if pricing were to be introduced in the near future, this would have to be done in a gradual and progressive fashion so as to allow water users to take appropriate reactive steps to minimise financial losses.

Thus physical scarcities can be expected in some catchments before prices could rise to the level necessary to prevent them. This will call for the reallocation of present entitlements, with resultant benefits and costs and - possibly - the need for compensation payments. This is a typical situation where CBA is able to assist in decision-making, and would merit inclusion in a decision-support system such as the Integrated Catchment Management model that is being developed by the CSIR's Division of Forest Science and Technology.

In addition, in the medium term the adoption of a pricing strategy could lead to significant transitionary losses to many present water users. CBA could produce important insights into ways of mitigating these through piecemeal variations in the overall strategy.

Inter-catchment allocation.

Under a pricing regime, the water-scarcity criterion is internalised: substantial water-use sectors such as forestry and irrigated agriculture would automatically be directed to catchments where water is abundant and hence would be relatively cheap. Without pricing, this does not always happen because economic location criteria unrelated to water scarcity (for example, proximity to markets or convenient transport infrastructure) dominate decision-making.

Thus, for however long it takes before an effective water pricing structure is in place in South Africa, there would be merit in using an exogenous method to internalise the water scarcity criterion in the decision-making that precedes the allocation of water to these sectors. CBA provides such a method. Instead of the present ad hoc rules, such as that afforestation permits should not account for more than ten per cent of a catchment's mean annual runoff, it would consider the costs and benefits to the country of alternative sitings of forestry and irrigated agriculture. Therefore it would provide a temporary allocative approach as a precursor to, and consistent with, an ultimate full pricing policy.

Environmental impact.

It is to be expected that future water supply augmentation schemes in South Africa will need to be of ever greater scope. Accordingly, their political acceptability is likely to become increasingly dependent on a satisfactory showing in terms of the criteria of Integrated Environmental Management. It is not yet recognised fully in this country that CBA is a logical component of IEM. Recently, however, CBA was included in an IEM study that had been performed on the proposal by Richards Bay Minerals to mine titanium at Lake St Lucia, with dramatic results by way of increased decision-making insights. Following this demonstration of the potential power of the technique, it is probable that CBA will become a standard part of the evaluation of projects with major environmental impacts - including large water supply infrastructural projects.

Policy issues.

Finally, carefully tailored CBAs may be helpful in developing elements of public policy.

Droughts, for example, are sometimes dealt with by crisis management. Arguably, they could be handled at lower direct and indirect cost if contingency plans were drawn up and steps taken to avoid economic and social development paths that are exceptionally vulnerable to drought.

Similarly, much of South Africa's spatial development in the rural areas has in the past taken place, and is continuing to take place, in regions facing major water scarcities under current non-pricing practices. Pricing may offer a solution as far as the management of water used for commercial purposes is concerned. Obviously, though it cannot equitably be applied to the very poor communities that are expanding rapidly in these regions. Thus a valuable contribution to policy-making could be provided by a proper evaluation of the longer-term costs that could be imposed on the country if growth of this nature is permitted to continue.

Many more such examples could be quoted. Suffice it to say, however, that judiciously directed CBAs could provide an abundance of far-reaching policy insights.

C2. VALUING NATURAL RESOURCES

C2.1 INTRODUCTION[1]

Business people give more weight than do conservationists to immediate profits and values over possible future profits and values. As the economic value of a protected area is difficult to measure, the short term economic gains from exploiting biological resources frequently appear more attractive than the long term benefits of conservation.

Many business leaders and economists also assume that economic growth through technological progress will automatically raise average living standards in the future. So why should the current generation pay higher prices and taxes to benefit future generations who will be better off anyhow? Cost-benefit analyses have a built-in bias against future environmental protection and resource conservation because they weigh future benefits and costs lower than current benefits and costs.

Conservationists put greater emphasis on future value of resources. Another serous limitation of cost-benefit analysis is that many things we value cannot be reduced to rands and cents, for example scenery, a wilderness area or red data species.

Governments are presented with a difficult decision of determining the best way to use their natural resources. Should they be preserved? Should they be converted to alternative uses such as agriculture? How should these decisions be made?

In most cases, a traditional economic analysis would indicate that alternative uses of protected areas would provide better financial returns than the more modest direct returns from retaining an area in its natural state. Financial analysis can often be misleading. The object of financial analysis is to examine costs and benefits which are reflected in market prices - it excludes those elements which are not traded (Dixon and Sherman, 1990).

1 This chapter is based in part on a forthcoming PhD thesis by J. D. Holland entitled "A determination and analysis of preservation values for protected areas", to be submitted to the University of Natal at Pietermaritzburg.

Many of the benefits of conserving natural areas are both difficult to measure and are non-market goods. As a result the value of conserving, as opposed to developing an area, is often underestimated. This leads to the bias referred to earlier in this section.

This paper is aimed at showing how economics can be used to improve the decision-making process when cost-benefit analysis is used. It is to be hoped that it will moderate the apparent reverence that individuals have for benefit-cost ratios. In the United States "... there is considerable evidence that officially employed cost-benefit techniques are the servants, not the masters, of political and bureaucratic interests" (Lynn, 1989, p54).

Ideally, natural systems and environmental quality analysis and valuation should be an integral part of any planning or decision making process. In practice however, they are absent or, at best, present only as afterthoughts once plans and projects have been formulated, major decisions made, and projects implemented.

Only now in South Africa are environmental analysis techniques being looked at with some interest. For example, the Development Bank of Southern Africa is looking at means to incorporate environmental costs and benefits into decision making. It is hoped that in the future these inputs will be used systematically for economic analysis of development projects and programmes.

The discipline of conducting cost-benefit analysis means that priorities among projects will at least be addressed in a systematic rational manner. The use of cost-benefit analysis will probably increase in the future as competition for scarce financial resources becomes greater and decision makers try to adopt a rational basis for choice.

Although there have been rapid advances in environmental and resource economics and in environmental management techniques, primarily in industrialised countries over the last ten years or so, and although there have been major advances in project evaluation in developing countries, the strands have evolved separately. In South Africa one is beginning to see the first signs of these strands being pulled together to show how these techniques can assist in incorporating the dimension of environmental quality into development planning. Apart from the Development Bank there are other institutions like the Department of Water Affairs and a number of engineering consultancy firms initiating environmental impact investigations.

Environmental Impact Analysis is only one of the tools available to economists and conservationists to investigate the ramifications on the ecosystem of investment and development decisions. An overview of some of these tools is discussed in the following section.

While it is recognised that any project assessed will be unique in its own right, many of the tools used in environmental analysis overlap with the result that they have in common a number of major problems and issues.

C2.2 ENVIRONMENTAL BENEFITS AND VALUATION PROBLEMS

A number of approaches and techniques have been developed which offer some promise for enhancing natural resource and environmental management To some extent these approaches do overlap and include, among others, environmental impact assessments, benefit-cost analysis, multiple objectives analysis, systems analysis and optimisation models, and input-output analysis.

This chapter is primarily concerned with the cost-benefit approach. However, it is necessary to first discuss some important theoretical problems associated with this subject. These issues include property rights, the concept of value, and benefit estimation methods which have been developed.

C2.3 PUBLIC GOODS AND PROPERTY RIGHTS

Goods which enter an individual's utility function can be divided into three categories: pure private goods, quasi-private goods, and pure public goods. Pure private goods are those traded in a formal marketplace, with full property rights. A property right needs to have three characteristics: it needs to be specific (to say

Class of Goods	Characteristics	Examples
Pure private products	Individual property rights	Agricultural
	Ability to exclude potential consumers	Motor cars Financial services
	Traded freely in competitive markets	
Quasi-private	Individual property rights Ability to exclude potential consumers Not freely traded in competitive markets	Public libraries Recreation in parks TV frequencies
Pure public	Collective property rights Cannot exclude potential consumers Not traded in any organised market	Air visibility Environmental risks National defence

Source: Mitchell and Carson, 1989

TABLE C1: CLASSES AND CHARACTERISTICS OF GOODS

what it is), it needs to be enforceable, and it needs to be transferable. The process of buying and selling should enable consumers to truthfully reveal their preferences

for goods which they want to consume. Quasi-private goods are, according to Kopp and Portney (1985) similar to private goods except that they are not freely traded in the formal marketplace. An example would be a trout fishing permit in Natal. The price is not determined by demand and supply, but by the provincial authority (often below the market price). While the market is not determining the price at which these goods are bought, it is nevertheless still possible to quantify the units consumed by individuals. Pure public goods, such as national defence, have no specific, enforceable, tradeable property right attached to them - consumers cannot be excluded from them and there is no rivalry attached to them. Table C1 summarises the above mentioned categories.

The major distinction between pure and quasi-private goods is the difference between individual and collective property rights. Rights which are held collectively occur where access to a good is held by all the members of a collectivity and individuals are unable to sell their access rights. If the good costs something then this cost is generally shared by all the members. The greater the excludability of the public good, the more the likelihood that the collective owns all the property rights. Water quality is an example of a good to which individuals have collective, non-transferable property rights of this nature.

A levy paid by flat owners is an example of a public good. The buying of the flat attaches private property rights to the flat, but the owner is also legally obliged to pay a levy each month, whose level is collectively determined, to maintain the block of flats. All the flat owners possess rights to use this non-divisible, collective good by agreeing collectively on the amount required by owners.

A collectivity sometimes grants individuals exclusive rights to consume a public good because the public is deemed to benefit from such an action. Usually those goods granted to an individual are excludable and open to congestion resulting in negative externalities. In this case allocation mechanisms are introduced such as auctions, first come - first served, or experience. Sometimes, as with mineral rights on public land, the rights are transferable. In these cases the transfer from public land is not exclusive, so that the government still has an interest in the property right and can rescind it. More common is where a collectivity grants a non-transferable right. Rights such as these are given freely to users of game reserves through the allocation of permits. Other rights can be allocated through fees to people who want, for example, to attain mining or fishing rights. The contingent valuation method is very well suited to measure the benefits attributed to quasi-private and pure public goods.

In summary, a pure public good can be described as a good which can be utilised by all consumers equally and, once it has been produced, it can be supplied to society at zero marginal cost. Consumption by one person has no adverse effects on the utility function of others. Furthermore, public goods are non-exclusive, so that individuals cannot be excluded (without cost) from the consumption of a public good once it has been produced. The level of externalities involved in the production and consumption of goods and services also has an effect on the degree to which a good is public. Environmental quality has a large degree of publicness, for once provided to an individual it can be utilised by others at zero cost.

Protected wilderness areas such as the Kruger National Park, are regarded as quasi-private goods (because of temporary individual property rights and excludability), but with public good attributes like scenic beauty. The public good attributes or characteristics lead to market failure, whereby private market systems are not able to provide Pareto Optimal price and quantity. As a result of this, there is a need to develop techniques to estimate demand for environmental quality to help frame public policy decisions.

The demand for a private good which is calculated from maximising utility, yields a consumer surplus as a welfare proxy. However, in the case of a public good the derivation of a demand curve and its associated consumer surplus is not as easy.

Public goods - similar to common property goods - are collectively consumed, but consumption by one member of the group does not lead to a subtraction of any other member's consumption of that good. All benefits are indivisible (Brown, et al., 1986, p. 27 ; Herber, 1975, p. 29 - 32).

Pure public goods are non - rival in consumption and are both non - excludable (for either technical reasons, or prohibitively expensive if exclusion is technically feasible), and non - rejectable. An example of a pure public good is the supply of national defence. All members of society have equal access to the good. The consumption of the good does not subtract from any other consumer's consumption of the good (non - rival in consumption), and if they live within the boundaries of the country they cannot reject the good (even if they are declared pacifists). The good is not divisible amongst consumers. (Herber, 1975, p. 32, p. 36)

For a pure public good, the marginal cost of an additional unit of consumption is zero since the good can be consumed by additional individuals without an increase in production costs (this is the extreme of the decreasing marginal cost phenomenon). The allocation of a pure public good in an optimal way requires that its price equals zero. The presence of both zero marginal costs and collective consumption where exclusion is not feasible provides a situation where public sector allocation is advisable. In addition, because of the large size of the group in the national defence example, individuals will be motivated by the free rider phenomenon. For this reason, voluntary private sector financing will be lacking and the public sector will have to intervene. (Herber, 1975, p. 36)

If exclusion was technically and economically feasible (the cost of exclusion escalates as the group increases in size) and politically acceptable, exclusive property rights could be defined and enforced, and the goods could be optimally provided by the market or by the public sector at an optimal charge. (Randall, 1987, p. 176 ; 5,8, p. 35)

Examples of pure public goods from natural resource economics include: ambient air of a given quality - each member of a large group can breath this air without reducing the amount available to others; the utility gained by an individual merely from the knowledge that a formerly endangered species has been re-established will not lessen the utility gained by others from the same knowledge. (Randall, 1987, p. 169)

Goods and services with pure public good characteristics are not prevalent in market economies. However, many "mixed goods" exist which have both private and public good features. Figure C1 which is adapted from Brown and Jackson (1986, p. 30) serves to illustrate four categories of goods.

Rival/ excludable goods or private goods may be provided by the market. These goods are divisible - a consumer can be prevented from consuming the good if he does not voluntarily pay for it. If the markets are perfectly competitive, Pareto efficiency in their provision may be achieved.

Rival/ non - excludable goods or common property goods. The benefits of these goods are collectively consumed but are subject to congestion. The provision of these goods is inefficient if left to the market unless exclusion is made possible by the specification and enforcement of non - attenuated property rights. The public sector can intervene to improve the allocation of these goods or they can be provided by private philanthropy.

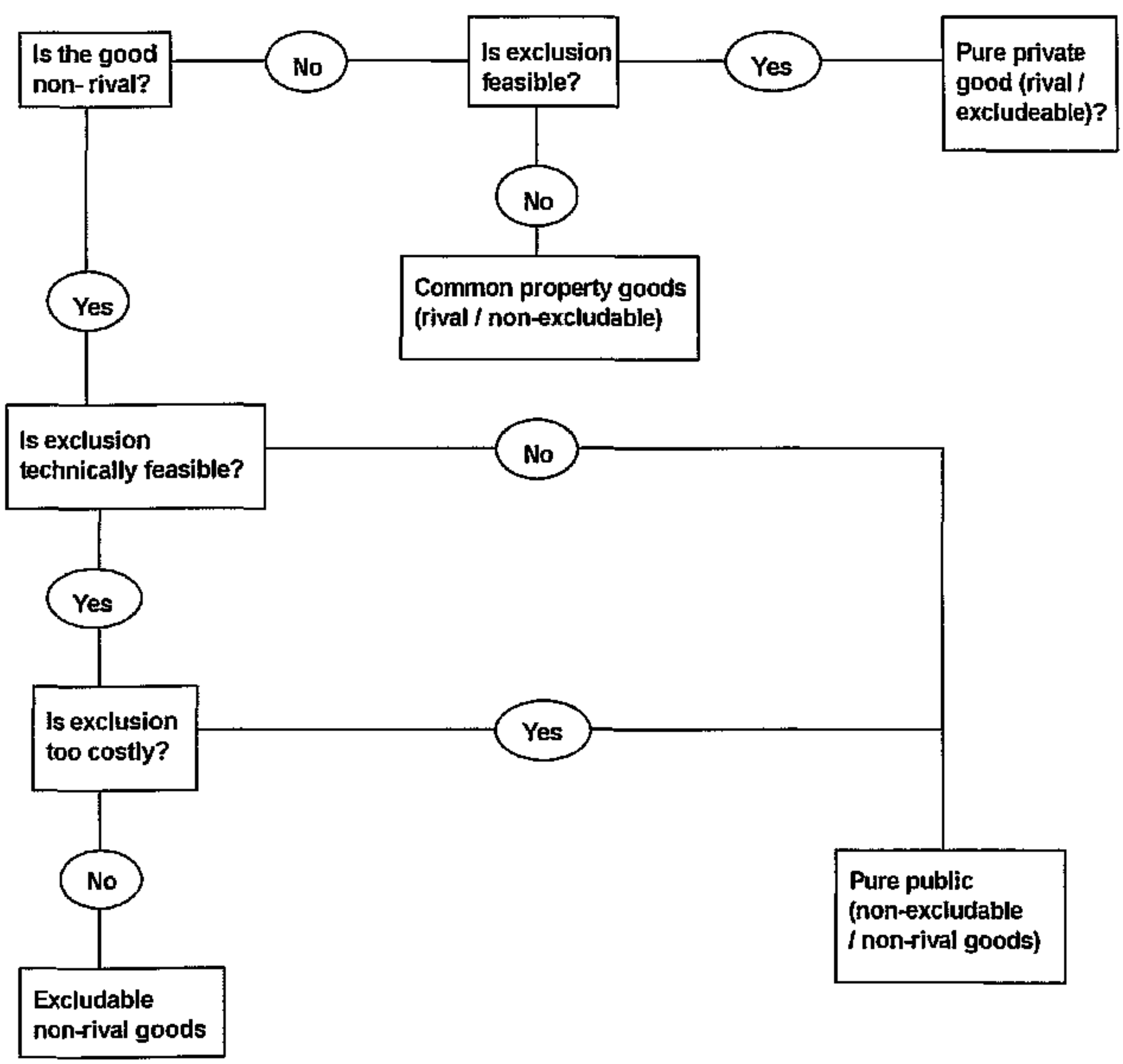

FIGURE C1: EXCLUSIVENESS AND RIVALNESS OF CONSUMPTION

Non - rival/ excludable goods. These goods can be provided either by the public sector at a user cost (for example, uncongested roads and bridges, inoculations or by the private sector at a determined price (for example, theatre performance, football match).

Non - rival/ non - excludable goods or pure public goods. These goods are provided by the public sector and their provision is financed by the exchequer.

There is a class of good which alters from non - rival (up to a specific number of consumers) to rival (over and above that specific number). This good is termed a contestable good. Over a specific range of users, from zero to X, additional consumers can be added without any rivalry. The marginal cost of adding additional users over the range will therefore be zero. The addition of users beyond X, i.e. X + 1, will result in congestion and the reduction of utility for all users. The marginal cost of adding users outside of the range (0 - X) will begin to rise and will approach infinity as an absolute capacity constraint is reached. Goods which can be consumed simultaneously by a number of individuals but are subject to a capacity constraint have the characteristics of a contestable good. Examples include many environmental amenity goods such as parks, scenic view points and hiking trails (Randall, 1987, p. 176).

The concept of value

The value of any public good consists of four components:

a. User value, this is the benefit derived by the consumer in present and future use.

b. Option value, which reflects values attached to future uncertainties. People may be willing to pay a premium to ensure their option of using a public good in the future. Two types of option values are presented as being potentially relevant for determining the demand for non-market goods or services; firstly, option value as a risk premium, and secondly, "quasi-option value".

- The risk premium option demand assumes that people are willing to pay an extra amount above the expected value of consumer's surplus merely to keep their options open when their future demands are uncertain.

- Quasi-option value is based on the principles of cost-benefit analysis. If new information about future benefits and costs will be available at a future time it is important that the relative reversibility of preserving natural environments be considered explicitly in the analysis. The analyst must not neglect the fact that choice of the development alternative has irreversible consequences. On the other hand, selecting a preservation alternative now does not bind society indefinitely to the preservation alternative and it is therefore possible to benefit from new information in the future.

c. Existence value, which is a value people attach to the knowledge that a good will exist in the future. Bear in mind that they may never see or use this good. Existence values represent the willingness of people to contribute towards the preservation of some natural resource apart from any plans on their part to receive benefits or enjoyment of the resource. For example individuals may be willing to pay something just for the knowledge that the hump-backed whale will continue to exist even though they may never see any in the future.

d. Bequest value, which is the willingness of a person to pay for endowing future generations with the same good. The aggregate of these four values is called preservation value (see Walsh et. al., 1984; Krutilla and Fisher, 1975, Mitchell and Carson, 1979; Brookshire et. al., 1983 and Brown, 1984).

Another important concept when dealing with natural resources is the concept of intergenerational transfers. One approach to account for intergenerational equity in cost-benefit analysis would be to lower the discount rate when dealing with the benefits which would accrue to future generations. Another may be to assign shadow prices to reflect interests of later generations. However, the complexities of intergenerational equity are not understood well enough to justify any adjustments to cost-benefit methods to reflect interests of later generations.

C2.4 NON-USE VALUES

Non-use values constitute a bundle of preservation benefits which Krutilla (1967) and Weisbrod (1964) termed option, existence and bequest values. Option value refers to the payment of a premium to retain the option of future use of a good, in addition to expected consumer surplus.

Option value is the difference between expected consumer surplus of use, and option price. This is, in turn, the maximum amount consumers, under conditions of supply or demand uncertainty, are willing to pay for the option to have a resource available for consumption in each subsequent year in which payment is made. Existence value is the WTP for the knowledge that a natural system or protected area exists, even though no consumption of the good is anticipated. Bequest value represents an individual's WTP for the knowledge or satisfaction gained from endowing future generations with protected areas. It is recognised that the concept of a bequest value can create uncertainty when separated from option and existence values. When combined into preservation values the concept can include option, existence and bequest values without problems (Greenley, et. al., 1981, pp657-672).

The literature suggests that option values are significant under conditions of uncertainty and will be positive for risk averse individuals demanding irreplaceable environmental systems. Bequest values depend upon uncertainty regarding a future generation's demand/supply for wilderness areas. Existence values appear to be related to the uniqueness and sustainability of the system, but do not have to be irreplaceable (Walsh, et al., 1984).

Being non-market public goods, preservation values are both non-rival and non-exclusive. Bradford's work (1970) in developing a theoretical foundation for CVM of determining an aggregate benefit function for public goods was extended by Brookshire et. al. (1980) To a general model for valuing natural resource flows, including natural systems protection. The aim is to determine a total value function which represents a person's ranking of alternative levels of WTP and protection of wilderness areas. Individual total value functions have the same economic meaning as indifference curves - the slope of which represents the marginal rate of substitution between income and WTP for protected areas (Hufschmidt, et. al., 1986, p68).

1. Recreation/tourism
2. Watershed protection
    - Erosion control
    - Local flood reduction
    - Regulation of stream flows
3. Ecological processes
    - Fixing and cycling of nutrients
    - Soil formation
    - Circulation & cleansing of air & water
    - Global life support
4. Biodiversity
    - Gene resources
    - Species protection
    - Ecosystem diversity
    - Evolutionary processes
5. Education and research
6. Consumptive benefits
7. Nonconsumptive benefits
    - Aesthetic
    - Spiritual
    - Cultural/historical
    - Existence values
8. Future values
    - Option value
    - Quasi-option value

Source Dixon and Sherman, 1990

TABLE C2: CLASSIFICATION OF BENEFITS

decisions made by managers concerning the allocation of public land to protected area status are based upon natural science criteria and the use-value consumers reveal. Even so, the gross expenditure methods used to calculate the latter are an understatement of demand. One can therefore question whether the benefits of additional protected areas exceed opportunity costs. The determination of economic values for protected areas should help with the formulation of sound natural environmental policies. Further, economic analysis cannot place a financial value on unknown ecological effects which again suggests that the total values calculated may be conservative.

C2.5 PROTECTED AREA BENEFITS

Different protected areas are associated with different benefits, depending on the type of areas and the conservation objectives. Some benefits result from direct use and are therefore valued in the market place, for example fishing. Other benefits, like recreation, are a function of people consuming or using protected areas. Most benefits, as we have noted, are difficult to measure in monetary terms.

A classification of benefits that makes it easier to discuss ways of valuing benefits is presented in Table C2.

Expanding on Table C2, Dixon and Sherman (1990) have highlighted the following points.

> RECREATION/TOURISM. Recreation and tourism are normally the primary objectives in national parks and are key objectives in many other types of protected areas. Unless the primary objective is strict protection of natural conditions or research, some tourism and recreational use are normally allowed. These services not only yield direct financial benefits from protected areas but stimulate employment and rural development in surrounding areas as well.
>
> WATERSHED PROTECTION. Maintaining the natural vegetative cover helps control erosion, reduces sedimentation and flooding downstream, and regulates stream flows. The extent of the benefit depends on the type of soils, topography, and natural cover in the protected area, the alternative uses available, and the types of investment and land use downstream.
>
> ECOLOGICAL PROCESSES. In their natural state, protected areas provide a number of environmental services in addition to watershed protection. These services often benefit people downslope and downstream by maintaining the productive capacity of nearby areas. Vegetative cover acts as a natural filter to reduce air and water pollution and promotes nutrient cycling. Clearly forests and wetlands are essential to the overall global life support of the planet. Many aquatic species depend on the existence of wetland areas during some portion of their life cycle. Mangroves and their associated fish and shrimp populations constitute just one example.

BIODIVERSITY. The maintenance of biodiversity - short for biological diversity, which includes all species, genetic variation within species, and all varieties of habitats and ecosystems - is currently considered to be one of the most important benefits of protecting natural areas. Biological resources from the basis of numerous industries and are major sources of food, medicines, chemicals, and other products used in both traditional and industrialised societies. By protecting habitats, one protects the variety of species they contain. For detailed discussions of the value of biodiversity, see McNeely (1988) and Wilson (1988).

EDUCATION AND RESEARCH. Research in protected areas may focus on a wide variety of topics from animal behaviour to measurement of environmental status and trends. By examining ecological processes in their natural conditions, one can better understand the workings of the environment and thereby improve management and restoration of both undeveloped areas and areas converted to other land uses. Research may involve changing the underlying conditions of the study area in some manner, or it may simply monitor natural conditions with as little interference as possible. Research is often integrated with education, as well, and protected areas provide fertile ground for field study by students at all levels, Moreover, protected areas provide fertile ground for field study by students at all levels. Moreover, protected areas instil people with an understanding and appreciation of the environment - making them more aware of the harmful consequences of certain types of behaviour.

CONSUMPTIVE BENEFITS Protected areas can yield a number of products including timber, forage, food, wildlife, fish, herbs, and medicines. If an area is to be protected, of course, such products will be harvested only on a sustainable basis. Depending on the objectives of the protected area, consumptive use of the resources may be totally forbidden (as in strict nature reserves and many national parks) or it may be a primary function (as in multiple-use areas).

NONCONSUMPTIVE BENEFITS. These benefits include the values people derive from protected areas that are not related to direct use. Aesthetic benefits may accrue when one passes near the area, views it from a distance or sees it in films or on television. The cultural value of a mountain or lake may be important in some societies, while urban societies may derive spiritual value from having a nearby asylum from modern life. Certain protected areas may also be key historic sites. Some people, moreover, may derive a benefit simply from knowing that a certain unspoiled area or a certain species exists, even though they themselves will never see or use it. This "exists, even though they themselves will never see or use it. This "existence value" is independent of any direct present or future use.

FUTURE VALUES. Apart from the values people derive from both consumptive and non-consumptive uses, the protection of certain areas ensures a variety of benefits from their potential use in the future - either for visiting or from products that may be developed from the area's genetic or other resources.table (Ledec and Goodland).

C2.6 BENEFIT ESTIMATION METHODS

Over the years a number of methods to calculate the demand for nonmarket goods and services have been explored. The analytical techniques for determining demand prices for public goods may be grouped into the broad categories discussed in this section (Mitchell, 1989):

Techniques to measure the benefits of public goods vary considerably in both their data requirements and the assumption with respect to the environment they operate in. Two categories of benefit measurement methods may be identified, these can be based upon: physical linkages and behavioural linkages.

	Direct	Indirect
	OBSERVED/DIRECT	OBSERVED/ INDIRECT
Observed market behaviour	Referenda Simulated markets Parallel private markets	Household production Travel-Cost Actions of bureaucrats or politicians
	HYPOTHETICAL/ DIRECT	HYPOTHETICAL/ INDIRECT
Response to hypothetical markets	Contingent valuation Allocation game with tax refund Spend-more-save-less technique	Contingent ranking Willingness-to- (behaviour) Priority evaluation survey question Conjoint analysis

Source: Mitchell and Carson, 1989

TABLE C4: BEHAVIOUR-BASED METHODS OF VALUING PUBLIC GOODS

The valuation methods in Table C4 are classified according to how preferences are shown and the type of behaviour linkage. The result is four categories of behaviour - centred techniques to estimate benefits.

OBSERVED/DIRECT TECHNIQUES. With this category of techniques, preferences are given in observed markets with the measures linked directly to the choice of consumers.

This set of methods is of limited value because the conditions under which they work need to be optimal. These methods are considered useful, however, to validate the measures calculated from the other three categories.

OBSERVED/INDIRECT TECHNIQUES rely on market choices made by consumers to supply the data, for example, driving to a resort or buying a good. The value of non-market goods must be inferred from market data which the consumer has expressed for another good which is indirectly linked to the good being assessed.

HYPOTHETICAL/INDIRECT TECHNIQUES. In this category the market is hypothetical and people's responses are indirectly related to valuing the good being considered.

Willingness-to-(behaviour) and conjoint analysis can be viewed as hypothetical analogues to the observed/indirect techniques discussed above. All indirect techniques observed and hypothetical, can be viewed as two-step procedures.

Instead of being asked to place monetary values on, for example, hypothetical trout fishing waters, people are asked how far they would travel to get to them. The contingent ranking (CR) technique requires the person to rank his preferences according to the different descriptions.

The researcher then translates the actions or responses into implied monetary values.

Another technique which may be employed is conjoint analysis. The objective of conjoint analysis is to enable a researcher to obtain a person's overall evaluations of a set of objectives or concepts and break these down into separate scores, called utility values or part-worths, for the various attributes of the objects which influenced the person ranking them the way he did. The technique is designed to expose a decision maker's utility functions that determine his preferences for alternatives among the set of multi-attribute choices he is faced with. This can be accomplished if the attributes, levels of choices and the respondent's priority ranking of the alternatives are known (Page, 1987).

HYPOTHETICAL/DIRECT TECHNIQUES directly measure the value people attach to hypothetical changes in the quantity/quality of goods provided. Smith and Krutilla (1982) conclude that the analyst's assumption that individual responses to hypothetical circumstances or transactions are completely comparable to individual responses revealed in actual transactions. These, they maintain, are institutional linkages because the organised markets in which the goods and services are traded are institutions that provide the information on individuals' marginal valuations of the commodity involved.

The spend-more-less survey approach is derived from survey instruments which ask people to say whether "we" (the country) allocate too much, too little or correctly for certain projects or programmes. The weakness of this method is that it elicits superficial and uninformed answers. Markets are not elaborated upon and even current expenditure patterns are not divulged to the respondent.

Allocation games that offer tax refunds are another method of estimating benefits. Instead of the person allocating a budget (allocation technique) among goods, the respondent is allowed to decline payment in favour of a tax refund for the public good being investigated. This encourages the person to value his utility for several goods simultaneously rather than, as in the contingent valuation (CV) method, individually. The technique involves superficial descriptions of public good categories, however, and WTP values elicited are not maximised.

The CV technique is based upon survey instruments designed to determine people's preferences for public goods by eliciting their willingness to pay (WTP) for specified improvements to the goods. It presents people with a hypothetical market where they are able to 'buy' the good in question. This technique is called the contingent valuation method.

A contingent valuation survey instrument consists of three elements. Firstly, a clear description of the good being value and the hypothetical situation in which it is presented to the respondent. Secondly, questions are put to the respondents to elicit their WTP for the good under question. Finally, socio economic data as well as data reflecting their attitudes to the good is collected. This data is incorporated into multiple regression equations to determine a valuation function for the good. 'Successful estimations using variables which theory identifies as predictive of people's willingness to pay are partial evidence for reliability and validity' (Mitchell and Carson, 1989, p3).

In table C5 a list of alternative approach or listing of benefit estimation techniques. It should be noted that the points presented in the tables are not exhaustive but at least give one with a point of departure.

Valuation technique	Benefits
Change in productivity	Watershed values Ecological processes
Loss of earnings	Ecological processes (health impacts)
Opportunity cost	Ecological processes Maintenance of biodiversity Global life support
Property value	Aesthetic
Wage differential	Aesthetic
Travel cost	Recreation/tourism Cultural/historical
Bidding games	Aesthetic Spiritual
Take-it-or-leave-it experiments Trade-off games	Cultural/historical Recreation/tourist Ecological processes Option value
Costless choice	Existence value Global life support
Preventive expenditures Cost-effectiveness analysis	Watershed values Maintenance of biodiversity Watershed value Ecological processes
Replacement cost/ shadow project/ relocation cost	Watershed values Recreation/tourism Maintenance of biodiversity Ecological processes

Source Dixon and Sherman, 1990

TABLE C5: WAYS OF VALUING VARIOUS BENEFITS

Cost-Benefit analysis (CBA)

Among non-economists, "cost-benefit analysis" and "cost effectiveness analysis" are often erroneously considered to be "techniques" for appraising public projects. If cost-benefit analysis (CBA) is to be considered a "technique", it is at best a loosely defined one. A costflow chart effective analysis is considered to be a special form or subset of CBA distinguished by the difficulty with which project benefits can be identified in terms of money.

Cost-benefit analysis is defined as an estimation and evaluation of net benefits associated with alternatives for achieving defined public roles.

The most popular criteria for choice in CBA is the net present value criteria for benefits or costs. The net present value (NPV) method uses a 'discount rate' to reduce a stream of costs and benefits, which are projected to occur in the future, to single numbers which can be compared. For example, if a product is expected to yield a benefit worth one hundred rands next year, we might value that one hundred Rand next year as, ninety five rands today. There are several reasons for discounting as well as a number of competing arguments as to how the discount rate ought to be determined, these will be discussed later. The net present value formula is:

$$NPV = \frac{Bo - Co}{(1+d)o} + \frac{B1 - C1}{(1+d)1} + \frac{Bt - Ct}{(1+d)t} + \frac{Bn - Cn}{(1+d)n}$$

Where Ct = Value of costs incurred in time t

Bt = Value of benefits incurred in time t

d = Discount rate

n = The life of the project in years

The main problem associated with the net present value method is how to determine an appropriate discount rate. Naturally, the higher the net present value of a project the better it is.

The internal rate of return (IRR) criterion is an acceptable alternative but not as popular as the NPV criteria. The IRR of a project is defined as that rate of discount of the future that equates the initial cost and the sum of the future discounted net benefits. Alternatively, it is the rate which would make the NPV of the project equal zero.

It is relatively easy to do a CBA when market related value of goods and services are on hand. On the other hand when the effects are outside of the market system it becomes more difficult to measure benefits and costs. In general however, there has been some success in developing useful, if somewhat imprecise models to estimate the value of things like outdoor recreation, pollution and safety. The concept of determining the option demand for goods and services which occur outside the market system is of particular relevance in this regard.

Another important concept when dealing with natural resources is the concept of intergenerational transfers. One approach to account for intergenerational equity in cost-benefit analysis would be to lower the discount rate when dealing with the benefits which would accrue to future generations. Another may be to assign shadow prices to reflect interests of later generations. However, the complexities of intergenerational equity are not understood well enough to justify any adjustments to cost-benefit methods to reflect interests of later generations.

Once a cost benefit project has been defined the next major problem is trying to identify costs and benefits. These may be classified in several ways, and classificatory schemes have been devised which include internal and external effects, incommensurable and intangibles, and direct and indirect effects.

Internal benefits accrue directly or indirectly to the study. For example the benefits from a private investment would be the revenues produced. External effects are more difficult to define and are those which "escape" the project and that fall into the hands of others. Sometimes these may be valued but cannot always be priced. For example, a hydro-electric dam constructed by the private sector may render flood control benefits to outsiders living downstream - these are external benefits. But a dam constructed by the Department of Water Affairs on behalf of the country may render flood control benefits to citizens of the country. To avoid undue controversy, externalities should be defined with reference to the project itself, and a thorough analysis should question whether the benefits can be captured, priced, and sold by the project entity.

External benefits therefore could be defined as those benefits received involuntarily by others for which they pay nothing. External costs are similar, being those costs imposed on others without compensation. Collectively these external effects are called externalities and they are neither deliberately produced nor deliberately consumed. An example of this may be the positive externalities resulting from research breakthroughs in the Nylsvley research project which may have a beneficial impact on grassland management in other parts of the country.

Externalities, incidentally, may be classified as technological or pecuniary. Technological externalities should be accounted for in a cost-benefit analysis - they are real and they increase or decrease social welfare. Pecuniary externalities normally should be excluded - they most likely represent redistribution of income, and their inclusion would represent double counting. It should be pointed out that a major stumbling block in identifying costs and benefits is the double-counting problem. Much of the criticism against CBA's is concerned with the counting of benefits more than once, usually in an attempt to cover all possible objectives for a project.

Incommensurable are effects which cannot be easily translated into the common denominator that is being used in the project. Intangibles are incommensurable that are not measurable in even their own terms. Examples of incommensurables include human life, air pollution, noise, scenic sights, recreation facilities, national parks, or even prestige.

With regard to direct verses indirect effects, a direct benefit of a product would simply be an increase in the real value of output associated with the project. This, most commonly, would be greater physical production. For example more power from a hydro-electric dam, purified water from a new dam, etc. Direct benefits may also arise from changes in quality, for example cleaner water.

Secondary or indirect benefits reflect the impact of the project on the rest of the economy. Secondary benefits are a form of external benefit. Benefits stemming from a project include the net incomes of processes between the primary cost and the consumers for example, transport contractors, builders and the like. This notion is similar to the idea of "forward linkages" used in development economics.

C2.7 CONCLUDING COMMENTS

The last few years have seen an explosion of interest in environmental problems amongst citizens of both developed and developing countries. Most of this interest has been focused on domestic problems and on possible changes in domestic policies designed to provide remedies. This section will look at a number of key problems, namely: prices, risk and uncertainty, and irreversibility.

Prices

Concepts which need to be understood before the connection between environmental problems and the economy becomes clear include the role of prices in allocating resources, the damaging environmental consequences of the free use of valuable resources that as yet are unpriced, and the manner in which these resources can be given prices. In a market economy, prices perform the key function of allocating all types of resources - raw materials, goods and services - to their most efficient use. When the market and the economy are functioning properly, the price each is resource can command is equal to the value of other resources that are used in producing it. In an economically efficient market it is not possible to produce an additional unit of a good without reducing the production of another good. One individual cannot be given more of any good without someone else getting less (the Pareto rule).

However, many environmental resources are still unpriced and remain outside the market. Because ownership rights have not assigned to them, and because they are not easily broken up into units that can be bought and sold, such valuable environmental assets as river systems, nature reserves, landscape features and even silence are "consumed" but their use is not accurately reflected in the price system. Economists describe the harms caused from such use as "externality," because the burden of the resources consumed falls on society at large, not just on the user who actually consumes them.

Usually such resources are consumed on a first come, first serve basis. Industrial air pollution spoils clear, breathable air; upstream polluters preclude downstream users; noise transportation and construction crowd out silence; and discarded cooldrink cans litter a community's open spaces. It is true that joint, non-exclusive uses may sometimes be possible. But such common property resources as clean air, open spaces and even sunlight are increasingly scarce because of usage that do not take into account the fairness and overall social desirability of the choices made.

Risk and uncertainty

Our understanding of the nature of environmental impacts and the value people place on them is somewhat imperfect. Indeed, it is difficult to think of any other area of public policy where there are comparable degrees of uncertainty. We have mentioned a number of situations in which uncertainty complicates evaluation, however, there are two main types of uncertainty; firstly technical uncertainties, because of imperfect scientific knowledge and secondly social uncertainties because of the difficulties of measuring social values attached to the environment

Impacts are difficult to predict, mainly because of the complexity of ecological interactions. This complexity means that it is often very difficult to know what to look for in the way of consequences, even though, after the event, it may turn out that scientific knowledge is quite sufficient to explain what happened.

Uncertainties are greater where there are direct effects on people's welfare other than effects on production. Even if some measure of "willingness to pay" can be achieved there is always uncertainty regarding its reliability. Evaluators treat data on willingness to pay as uncertain and risky stuff, and rightly so. Perhaps the greatest difficulties arising from social uncertainty are associated with long-run impacts on the environment For example, consider the destruction of wilderness areas by industrial development at the cost of "recreational benefits". The problem here is that the relevant value people put on these benefits compared with other things they consume, may be expected to rise as time passes. If people put sufficient value on wilderness, there may be more radically conservationist policies in the future to preserve it. If so, the price of a portion of the wilderness will not rise as much as it might without the conservation policy. Or peoples' tastes may change as they substitute some other activity for utilising wilderness areas. Once again, the effect would be to hold back the increase in the price of wilderness relative to other things. There are deep uncertainties about these matters and even if we agree that the most likely outcome in the future is a rise in the relative valuation of wilderness areas, such as has happened in the past, we are hard put to say just how big any anticipated increase in prices may be.

Irreversibility

More than the usual degree of uncertainty surrounds the potential future benefits of conserving ecosystems. Irreversibility is clearly central to thinking about endangered species or ecosystems because extinction or loss of wild lands is indeed irreversible. Distinctions can be made among decisions and actions on the basis of whether their consequences are difficult or impossible to ameliorate. It can be argued that the wholesale loss of species or wilderness ecosystems falls into the category of consequences that are impossible to reverse and difficult to ameliorate.

With regard to endangered species, it should be noted that one of the valuable features of wilderness areas is the variety of natural populations they host. The conventional view of the threat to natural populations - endangered species - is that it is due to exploitation. In some cases, this is undoubtedly correct. But the major threat to biological resources is habitat modification. This can take several forms: direct conversion, as in the drainage of wet land, for the development of dry land, for agriculture, housing and transportation; chemical pollution, as from acid rain; and "biological pollution", the introduction of exotic species. Of these, the most important currently appears to be direct conversion for agriculture and other developments. Thus, the issue of endangered species protection is intimately related to that of wilderness preservation.

One of the issues that is difficult to convey is the question of how does preservation of flora and fauna populations not harvested now contribute to human welfare? In reply to this there may be, let us say, a discovery at the Nylsvley's savannah ecosystem project which may be an agricultural breakthrough with respect to a grass hybrid which would lead to a significant increase in agricultural output. This might appear highly conjectural, but the point is clear. Just one apparently trivial botanical discovery can result in an enormously beneficial spin-off for agriculture. More generally, this example (there are many others) illustrates one way in which a currently unharvested species can contribute to human welfare: by conserving genetic information that may be in the future more useful to some form of economic activity.

A second major way in which species are useful is as components of living ecosystems that provide the basic physical and biological support for human life. These include maintenance of the quality of the atmosphere, control and amelioration of climate, regulation of fresh water supply, generation and maintenance of soils, disposal of waste, and cycling of nutrients. Removal of any one species may cause a system to break down because each has evolved a set of characteristics that makes it a unique functional part of the system.

Loss of natural populations can also adversely affect human welfare in less tangible ways. We ought to at least note, though we can do little more than this, that some of the concern for endangered species is of a religious or ethical nature which does not easily fit in to our utilitarian framework.

The challenge is twofold. First, ways must be found to discriminate among areas tabled for conversion so that those richest in species can be afforded some measure of protection. Such an approach would recognise that some conversion will take place. The object would be to minimise the related losses. Second, ways must be found to finance the desired protection.

Why should there be more concern about disturbing scenic natural areas and endangered species' habitats than about issues involved in the allocation of resources of comparable value? The reason is, at least partially, because wild lands and natural populations are the result of geomorphologic and biologic processes that represent a time frame measured in eons and, thus, cannot be reproduced by man.

Kneese and Sweeney (1985) note that the distinction between reversible and irreversible decisions in economic processes can sometimes be illustrated by the differences we can observe between production and investment decisions. A producer with a given plant and equipment, inventory of raw materials, and stock of finished goods faces the expected demand which he intends to meet. His decision on level of output in each product line may not be entirely consistent with the actual demands, and these discrepancies will be observed by changes in finished stock inventories. As the errors to the level of the production required to meet the demand are encountered, stocks in inventory will rise (fall) to the extent of the over (under) estimate, and he can thus adjust output level of the production line to conform to the actual demand. While his original decision may not be rescindable for any given production batch, he can alter the consequences by adjusting production on subsequent production runs. In this sense, we can consider decisions reversible; that is, if the consequences of a decision can be readily altered with negligible losses, it may be likened to a decision that is reversible.

If the decision, however, relates to the capacity of his plant so that he is required to make decisions on the amount to be invested for its construction, the consequences of a poor decision will have longer duration. Investment in plant and equipment, unlike investment in raw materials, cannot be liquidated in any short period of time. Indeed if the capacity originally estimated to be required exceeded the market potential for his output, he would have made an irreversible commitment because capital is neither easy to modify nor readily liquidated. This, then, characterises one aspect of irreversibility in economic processes - the inability to recoup investment in excess capacity.

APPENDIX D

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D2. BIBLIOGRAPHY OF REFERENCES CONSULTED TO DATE — DECEMBER 1992

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