Task 1.3.2
Externalities

Valuing Externalities: A Methodology for Urban Water Use

John Bowers and Mike Young
Policy and Economic Research Unit

Urban Water Program

CSIRO Urban Water Program

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PREFACE

This draft report has been prepared under contract to the CSIRO Urban Water Program that is ably led by Andrew Speers. During its preparation, we have had the opportunity to work closely with the Program's Project Managers. The innovative work they are leading suggests that it is technically feasible to significantly improve urban water use. The technical opportunities are many and a significant number appear to be affordable. The focus of this report is on valuation of externalities.

Many people who supplied documents and background information have assisted preparation of this report. Their contribution is acknowledged with gratitude. In particular, we would like to thank Doug Young and Stefan Hajkowicz for their perceptive comments on the proposed method and Sharon Rochow for assistance with the preparation of this report.

The next step is to test the methodology in one or more locations. One obvious place to do this is the Ellenbrook case study. We caution, however, that many of the externalities in this area arise from agriculture.

John Bowers

Mike Young

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VALUING EXTERNALITIES: A METHODOLOGY FOR URBAN WATER

Introduction

The objectives of this report are to define and describe `externalities including resource management costs' for the urban water sector and to provide a method of valuing. From an urban water perspective, externalities are the sub-set of non-market effects on the welfare of third parties and the environment, that arise from water use. A more precise definition is provided later in the report.

The report is prepared against the backdrop of the Council of Australian Government's (COAG) Water Reform Framework. This framework proposes a transition to a situation where there is full cost recovery for water use. Full costs are defined to include externalities.[1] For the purposes of this report, we assume that full costs means full long-run marginal cost. In parallel with this process, the CSIRO Urban Water Program and others are seeking to find ways to improve urban water use. Two uses can be identified for the methodology and recommendations presented in this report:

- price determination; and
- assessment of the costs and benefits of alternative ways of managing the urban water system.

A parallel report (Young, 1999) identifies opportunities to use the proposed method and the information it provides to improve the nature of externalities associated with urban water use. This parallel report is practical in its orientation and written to reveal the nature of global experience in managing externalities.

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1 Section 3 (a) (i) of the Strategic Water Reform Framework requires the adoption of pricing regimes based on full-cost recovery. The Expert Group established under section 3 (b) (i) defined full economic costs as the sum of operating and maintenance expenses, administrative costs, replacement cost depreciation, the opportunity cost of capital and externalities (COAG, 1995). The Guidelines for Water Pricing, endorsed by all jurisdictions, broadly accepts this definition, but in Principle 5, clarifies the notion of the opportunity cost of capital and specifies the treatment of taxation in the calculation of full costs. Principle 7 requires transparency in price determination in the treatment of various items of full cost including `externalities'. Externalities are further defined to include resource management costs.

The urban water system

In most parts of Australia, the supply system draws its water from surface water but in a few systems use is made of groundwater (Figure 1). The water supply system and the storm water system are largely independent but interaction between them is common. An overview of the place of the urban water system in the Australian economy is provided in Box 1.

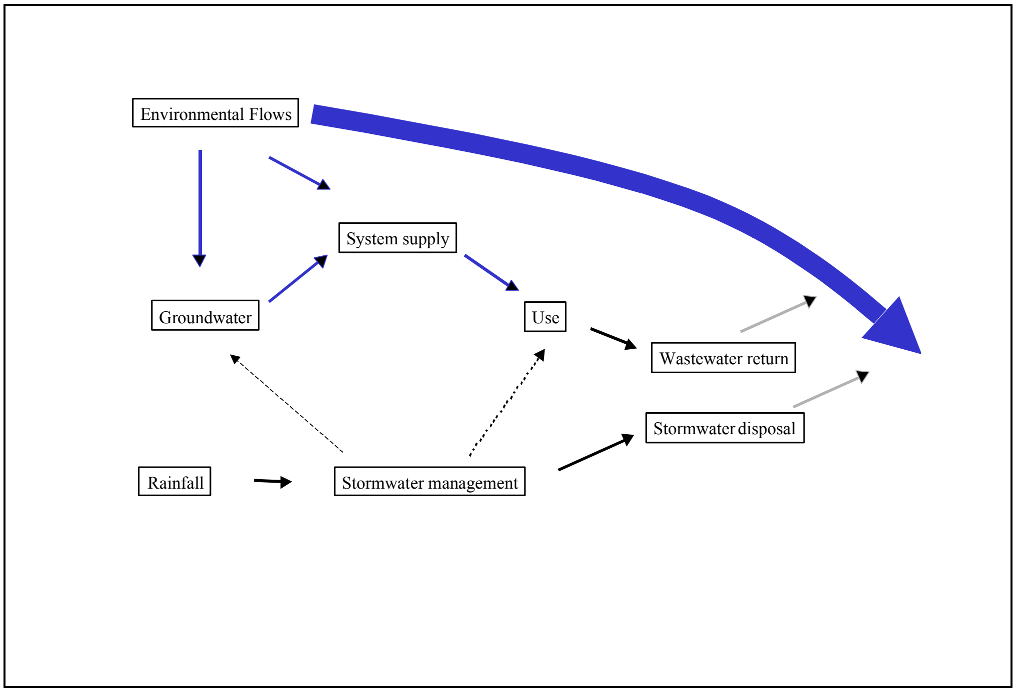

Figure 1 An overview of the urban water system

The complete urban water system consists of:

- water supply;
- a wastewater return;
- a stormwater (Box 2).

A description of each of these components is provided in Box 2.

The urban water supply system

The typical Australian situation has a single body, normally a public monopoly utility, responsible for both supply of clean water and treatment/disposal of dirty water to a set of customers. The customers demand clean water, some of which they convert to dirty water, that is returned to the utility for disposal. Thus, the flow of dirty water in the system

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is less than the supply of clean, the difference comprising system losses. System losses have two components: transmission losses and water consumption.

Box 1 An overview of the place of the urban water system in the Australian economy

Australia demands about 20,000 Gigalitres (GL) of water per annum. Most, 72% is for irrigation of agricultural and horticultural crops and pasture. Urban demand, including industries and services located in urban areas, is about 3,500 GL per annum. Urban domestic households demand 12% of the remaining 28%.

Urban water use has been growing at a little over 2% per annum over the last 15 years. But this growth in use is more than explained by the growth of the urban population. Per capita water consumption is declining in the urban sector and expected to continue. AATSE (1999) forecast growth in domestic demand at 1.8% per annum to reach about 4,000 GL by 2020 as a result of urban population growth. The forecast growth rate of urban water demand is lower than that of the rural sector.

Urban areas are responsible for almost all the flows of wastewater into both inland and coastal waters. Although they may contain contaminants such as nutrients and pesticide residues, return flows from irrigation systems are not classified as wastewater.

In aggregate, urban wastewater flows in Australia are currently slightly less than 2,000 GL per annum and are forecast to rise to 2,500 GL by 2020. Because most urban development is on the coast, the overwhelming majority of wastewater flows (about two thirds) are discharged to coastal waters.

Source: AATSE, 1999.

The destination of the wastewater determines the nature of the system. In a *sequential system*, wastewater is available for reprocessing as clean water. In the extreme case, abstraction and discharges are to the same water body so that dirty water discharges automatically contaminate the clean water supply. In less extreme cases, the sources are partially or wholly separable. The alternative *nodal system* is one where the discharges are wholly outside the supply system and cannot contaminate it. A coastal community with discharges to the ocean might be a pure nodal system. Because the majority of the population lives close to the coast, Australian systems are mostly nodal. This is not the case in other countries. In many parts of the world water is reused many times. The Murray

Darling River system is probably Australia's best known example of a sequential system.

Stormwater

In Australia, stormwater collection and disposal is typically separated from the wastewater flow. There is thus no need for the responsibility for stormwater to be vested in a single wastewater authority. Typically, stormwater drainage is managed by local government. In larger population centres often sewage and water supply is managed by an independent authority or company. This is not always the case elsewhere. In the UK, for instance, stormwater is normally channelled into the sewage system. The economic case for separation hinges on the variability of rainfall. In a dry country, such as Australia, integration would require an expensive increase in the capacity of the sewerage infrastructure to cope with irregular large flows. In practice, the choice is not dichotomous, and the optimal division of stormwater flows, between the sewers and dedicated infrastructure, is likely to vary across and between urban areas. Recent thinking sees a case for more integration than was thought desirable in the past (Thomas *et al*., 1997).

Regardless of whether economic and engineering factors favour integration or separation of wastewater and stormwater flows, stormwater management and disposal is an activity outside of the water supply system. Stormwater, in fact, imparts externalities to the water supply system, causing sewers and sewage treatment facilities to overflow and contaminating sources of water supply system. We return to this issue below.

Box 2

System supply modifies the flow of water through the environment to enable delivery to users. Modifications can be both temporal and spatial:
- Spatial - Impoundment (reservoirs etc);
- Temporal - In general water demand patterns and, hence, abstraction will differ from the pattern of system flows.
Normally, system supply will tend to reduce seasonal variations.

Use reduces the volume of water in the system and available for the environment, recreation, etc.

Wastewater return chemically modifies the water system. Some of the water supplied is converted to dirty water by the recipients. When returned, dirty water alters the composition of the system downstream of the point of discharge. *Stormwater management* has the potential to decrease or increase recreational and amenity values through sensitive design. Local wetlands, for example, can improve landscape amenity areas. Recreation fields can be built within floodways.

Stormwater *can be used to recharge aquifers. Stormwater disposal* erratically amplifies flows through the system and introduces pollutants into it from both external and internal sources.

Pricing water services

A producer of a good or service would be expected to set a price to cover the full costs of its supply and to earn a profit. Because of the essential nature of water services to human welfare and environmental health, other factors also feature in the pricing decision. These include issues of wealth and income distribution and the costs of resource management. But, in addition to cost recovery and distributional questions, water pricing may be used for a variety of purposes and it is important to clarify them. The principal ones are:

i. To internalise the externalities of specific uses of water;

ii. To manage or reduce overall demand by targeting specific uses; and

iii. To encourage water use practices that reduce the costs of water supply.

Failure to understand the multiple purposes for which water prices are used can lead to confusion. One reason is that the perception of externalities is situational or context dependant. Thus, in a context of excess demand for water (e.g. a drought), non-essential water use such as watering lawns or garden flower beds may be viewed as anti-social and there may be a temptation to present non-essential uses as imparting negative externalities on the system. In fact, in such a context, targeting non-essential uses is a perfectly legitimate and possibly effective means of demand management. It is a standard instrument for drought management in, for instance, the UK.

As another example, the separation of black and grey water flows in discharges from households, permits substantial reductions in the costs of dirty water treatment. Utilities could legitimately use the water pricing system to encourage consumers to invest in separation capacities and to adopt practices that lead to separation. There is then, a temptation to present the household that does not separate black from grey flows as conveying a negative externality on the system, raising the cost of dirty water treatment and thereby imposing costs on other householders and possibly on the environment. This is the case in purpose (iii) above.

Water use practices that raise costs of supply are only an externality if catchment management plans, or some other form of regulation that defines the structure of property rights, make them so. Thus, if a statute were adopted which required households to separate grey and black water, then the household that failed to do so would be conveying an externality on others.

If the COAG principles of full cost pricing are to be implemented it is necessary to clarify what is meant by an externality.

What is an externality?

In a market economy, individuals and households satisfy many, probably most, of their needs by buying goods and services 'in the market place'. The household's ability to buy goods and services, and, hence, to satisfy its requirements, is determined by its income, and in official statistics we use the income of the household as a measure of its level of welfare or standard of living.

Economists see the household's welfare as achieved by buying and selling in the market place. As such, the welfare levels of households are interdependent, since the prices of goods and services are determined by the forces of supply and demand. But trading in the market place and altering prices is not the only way in which economic units such as households and firms affect each other's welfare. They also do so directly without the intercession of markets. Where that happens we have a potential externality.

Definitions of externalities vary (Baumol and Oates, 1975). Most define them, as we do, as a legitimate action by one economic unit that impacts on the welfare of another economic unit that does not take place through markets (Bowers,1997). McTaggart *et al.* (1999) define an externality as a cost or benefit that arises from an economic transaction that falls on people who don't participate in the transaction. Not all non-market transactions are externalities. As Baumol and Oates observe, "If I purposely manoeuvre my car to splatter mud on a pedestrian I dislike, he is given no choice in the amount of mud he 'consumes,' but one would not normally define this as an externality." Anti-social forms of behaviour, crime, etc. are usually excluded.

One of the characteristics of an externality - lack of deliberate intent to harm - is important (Mishan, 1969). If I decide to burn down my neighbour's house there is no externality, but if sparks from my barbecue set it alight then there is. It does not matter for this definition whether, when I lit the barbecue, I was aware of the fire risk, nor whether my action was negligent. As long as the intention was not to cause a fire and no laws were broken, we have an externality. Similarly, when a factory or sewage treatment plant legally disposes effluent into a stream, it probably does not take into account the costs that this action imposes on members of a fishing club who like to fish lower down in the stream (McTaggart *et al.*, 1999).

Internalisation of externalities: making those who cause the externalities aware of what they are doing and providing them with incentives to change their behaviour; is an important objective of environmental policy. This is, of course, one of the reasons for valuing and pricing for externalities. There are, however, many ways of internalise externalities. Grants, regulations, information, tradeable emission permits and Pigouvian[2] taxes are all options (Young, 1999).

Externalities can be either positive - one economic unit's action raises the welfare of another – or negative - one unit's action reduces the welfare of another. The increase or decrease in the welfare is called the social benefit or social cost. This terminology is summarised in Table 1.

It is consistent with the definition of an externality to consciously design systems so as to maximise positive and minimise negative impacts. Recent developments in the design of stormwater collection and treatment systems have created a whole new range of positive externalities in the form of improvements to the urban environment. These include created wetlands and the return of flow channels to natural states. Since they require capital investment, the costs of these welcome improvements will be incorporated into the price of

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2 In a classic text, Pigou (1920) formulated the idea that by taxing decision makers, the party with the property right, externalities could be internalised.

stormwater services. These improvements are part of the duty of care that we discuss below.

Table 1. Terminology of Externalities

	Description of process	**Welfare change classified as**
Positive externality	Economic Unit A's action improves unit B's welfare	Social benefit
Negative externality	Economic unit A's action worsens unit B's welfare	Social cost

There are two approaches to the definition of what is a social cost and what is a benefit. The conventional approach is to define costs in terms of the current practice and current albeit incomplete, specification of property rights. The alternative approach is to measure benefits against an agreed benchmark.

Indeed, public perceptions as to what is a positive externality and what is a negative externality often differ to that defined by economists. The common lay interpretation of a 'negative externality' is anything that a person causing the externality should be charged for. That is, the polluter should pay.

In economics, typically, the frame of reference used is the status quo. Measured against the status quo, pollution reduction is defined as a 'positive' not a 'negative' externality because the action increases the welfare of those affected by it. The way out of this communication impasse is to develop new language which sends clear transparent signals.

Generally, use rights and obligations are defined in legislation, regulations and catchment management plans. Real property arrangements and license conditions also affect such definitions, as does common law. Collectively, all these mechanisms define each person's duty to care for the environment. **Duty of care for the environment is a term gaining gradual acceptance in rural areas. Duty of care is a new concept for urban water users.**[3] Often the implied definition of duty of care suggests some degree of social acceptance and tolerance of practices that degrade the

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3 The idea of duty of care for the environment was developed by Binning and Young (1998) and has since been given wider currency by the Industry Commission (1998).

environment. Water users are allowed to create some ‘negative’ externalities but not too many!

Acceptance of a duty of care benchmark means that any activity that produces an outcome above this standard is a positive externality. Duty of care, however, is not a static concept. In particular, management plans often propose to change the way duty of care is defined. Most catchment management plans suggest an expectation that standards will be raised through time and raised without payment of compensation. For example, the discharge of secondary treated sewage to many river systems was considered acceptable until algal blooms became common. Now tertiary treatment is required. **Duty of care is an evolving concept** (Figure 2). **Catchment Management plans can define duty of care as a set of minimum water quality objectives (WQO) that all users must pursue. This means that positive externalities may *initially* be most appropriately dealt with through the use of positive price signals but, at some stage in the future, be more appropriately managed via the use of negative price signals.** Further explanation is necessary to clarify this point. To speed adjustment, initially, people causing the externality might be paid an incentive to encourage them to change practice. After an appropriate period, however, all might be expected to adopt the preferred practice. When compliance becomes a duty, those who do not comply should be penalised via the imposition of levies, fines, etc.

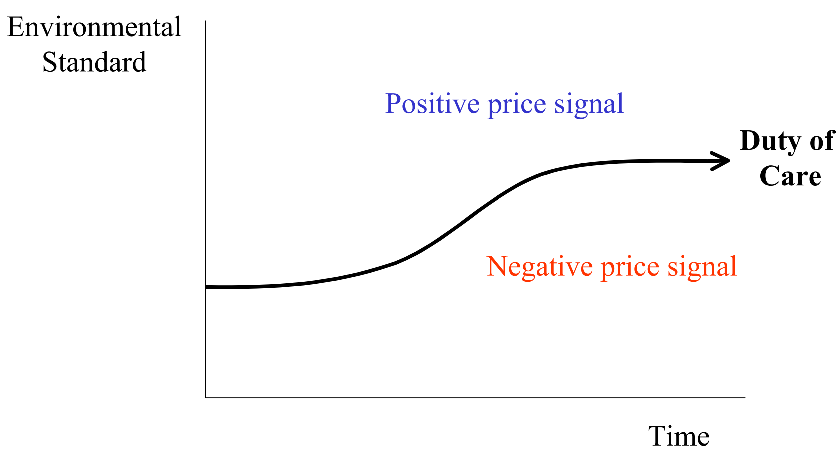

Figure 2 Dynamic nature of the distinction between positive and negative externalities. Actions defined as positive today may be defined as negative tomorrow

Tangible and intangible externalities

From a valuation perspective, there are two types of externalities;

- tangible externalities; and
- intangible externalities.

After the occurrence of a flood, society counts the cost. For many of the impacts it is possible to impute money values to the damage. These include such things as the costs of repairing buildings, redecorating and replacing furnishings and vehicles that have been destroyed or damaged beyond economic repair. Equally, the cost of clean up of pollution can be calculated – so many workers with the equipment for so many days. These impacts are known as *tangibles;* the assets effected, whether items of furniture or workers, are traded in markets and have market values.

With thought, the area of tangibility can be made quite wide. Thus, with people who have been injured or have died in the flood, we can calculate the hospital and funeral costs that have been incurred. If the casualties are children or adults of working age we can also impute their lost earning potential using market information about wages. This can either be seen as a loss of income to the affected households or lost productive potential to employers or society. But however inventive we may be, the tangible area has limits. There is no market that allows us to put a value on human life nor on the pain and suffering experienced by the victims or the grief of their loved ones. Pain, grief and suffering are *intangibles*. An intangible externality is one where there is no market to which reference can be made to achieve a valuation. Damage to wildlife and losses of biodiversity are intangibles; some amenity effects are intangibles also.

The key to distinguishing tangible from intangible externalities is that intangibles possess the characteristics of public goods (or public bads). (see appendix 3)

Classification of Intangibles

Valuation of externalities is simpler if a classification system can be found that is discrete. That is, that there is no overlap between any two categories and, hence, minimal risk of double counting. One way of doing this is to focus on the receptors where each externality expresses itself.

The externalities should be identified and classified into the following categories:[4]

- Production externalities;
- Property damage externalities;
- Fisheries externalities;
- Recreation externalities;
- Amenity externalities;
- Health externalities;
- Wildlife and bio-diversity externalities.

In Table 2, externalities are classified spatially and by economic service. This classification should be applied separately to each water service then quantified and valued using the approaches set out below and in Appendices One and Two. In many cases, it will be more efficient to value all the externalities in one area and then partition them according to their source.

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4 COAG Water Reform documents define externalities to include resource management costs. By this they mean the costs faced by governments as they monitor externalities and manage them to minimise negative impacts. Strictly, these are not externalities but rather direct costs of water management.

Table 2 Mapping of conceptual to spatial analysis of externalities

Type of externality	Spatial location		
	Upstream	Within system	Downstream
System Supply Externality	Wildlife (+/-) Fisheries (+/-) Recreation (+/-)	Wildlife (+/-) Fisheries (+/-) Recreation (+)	Wildlife (?/-) Fisheries (+/-) Property (-)
Use Externality		Production (-)	Wildlife (-) Fisheries (-) Recreation (-) Health (-) Production (-)
Wastewater Return Externality			Wildlife (-) Fisheries (-) Recreation (-) Health (-) Production (-) Amenity (-)
Stormwater Management Externality		Property (-) Recreation (-) Health (-) Production (-) Amenity (-/+) Wildlife (-/+)	
Stormwater Disposal Externality			Property (-) Fisheries (-) Recreation (-) Health (-) Production (-) Amenity (-)

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Valuation methods

As explained in detail in Appendix Two, different valuation techniques are appropriate for tangible and intangible externalities. Thus, it is necessary to classify the identified externalities of the water supply and the stormwater system by their tangibility.

Table 3 Classification of tangible and intangible externalities

Economic Service	Type of Externality	
	Tangible	Intangible
System Supply	Recreation; property	Wildlife
Use	Production; recreation (part); health (part)	Wildlife; recreation (part);health (part)
Wastewater Return	Production; health (part); amenity (part); recreation (part)	Wildlife; health (part); amenity (part); recreation (part)
Stormwater management	Production; health (part); property; amenity (part); recreation (part)	Wildlife; health (part); amenity (part); recreation (part)
Stormwater disposal	Production; property; amenity (part)	Wildlife; health (part); amenity (part)

A framework for valuation

The steps in the valuation process are as follows:

1. Identify the externalities separately for each water service (water supply, sewage treatment, and stormwater disposal) and classify them according to the above categories. Efficiency requires that the externalities attributable to each service should incorporated into the prices of each service (Box 3).
2. Determine the physical magnitude of each externality through the application of a dose response model. (Appendix One)
3. Value the physical magnitudes in the ways discussed below and summarised in Table 4. The valuation methodologies are discussed in Appendix 2.

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Box 3

Principles of water service pricing

1.Separation of services

Economic efficiency requires that consumers should be faced with a set of prices for services that reflect the relative costs of their provision. In the current context this means that the full costs of each water service, including externalities, as defined by COAG should be incorporated into the price of that service. This means:

Water supply should be metered and charged the full long-run marginal cost per litre.[5]

Sewage discharges, ideally, should be metered and separately priced. Realistically metering is not possible at present, so that sewage charges will be on a per property basis. None-the–less, the sewage charge should be clearly distinguished from charges for water supply and, where appropriate, storm water charge and should be based on the full long-run marginal cost of the service. Account should be taken of the possibility of system failure and impacts such failures may impose on tangible and intangible values of concern to people.

Storm water charges should be property based, in proportion to anticipated load and calculated on the long-run marginal cost of disposal. Where sewage and storm water disposal are provided by a single utility, the two charges should be clearly separated. It is desirable that indicators of contribution to the problem should differentiate stormwater charges. Indicators should be simple. Ideally charges should increase with roof area of property or, preferably, total impervious area. Deductions for installation of a storage tank and a functioning pervious soakaway system and a surcharge for a metal roof are options to be considered. The scaling factors should be based on estimates of how total storm water volume entering the disposal system varies with the area of impermeability multiplied by long-run marginal cost per unit volume of storm water. The deductions similarly should be based on estimates of the contribution that these factors make to the reduction in storm water load multiplied by unit volume full cost as before. Metal roofs add to the heavy metal content of stormwater that pollutes the receiving water and has adverse impacts on wildlife and fisheries and, if the receiving water is used for water supply, imposes treatment costs. The surcharge should be based on the cost of treatment necessary to keep the heavy metals out of the receiving water. Alternatively, and as a second best, the cost of meeting water quality objectives plus any identified costs of treating water for use should be used. The problem of stormwater pollution may additionally require policies directed at other targets, e.g. on the behaviour of car and pet owners.

Production, property damage and fisheries are tangible externalities. Recreation and amenity are partial intangibles. Health externalities are part tangibles and part intangibles.

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5 It is not clear from current statements by COAG whether or not full cost means full average cost or full marginal cost. We assume that they mean full long-run marginal cost as this is what is required for economic efficiency.

For this exercise, however, we recommend that they be treated as intangibles. This is for several reasons:

1. The imputation of tangible health effects (sickness and resulting loss of working time and death) to environmental pollution has proved extremely difficult even in relatively straightforward cases of air pollution.
2. In the water environment, public health restrictions on the composition and location of discharges already serve to minimise those impacts. The one possible exception is discharges to coastal waters but even here proven health effects are difficult to establish.
3. Water quality objectives are designed with an eye to human health as well as wildlife considerations so that pricing for water quality improvements benefits both.

The only exception to this practice should be stormwater impacts where injury and death from flooding are possibilities.

Wildlife and bio-diversity effects are pure intangibles.

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Once the physical magnitude of the externalities has been determined via dose response modelling the valuation of externalities is as follows.

For simplicity of presentation, the following discussion assumes that the effect of the externality is continuous. Where effects are periodic, as with stormwater, events and system failure, and also with externalities of low water flows and system failures, the valuation should be multiplied by the estimated annual probability of the effect occurring. The calculation of return periods and adjustment of costs and benefits for them is standard practice in engineering manuals on flood prevention, and these should be consulted where necessary.

Our proposals for the valuation of intangibles rely on WQO as expressed in catchment management plans, etc, which are discussed later in the report. In addition, and as stated earlier in the discussion about the role of catchment management plans and the importance of understanding duty of care, a person experiencing an externality may not have to pay to change it. The question of the appropriate instrument or instruments to use depends, among other things, on what rights they have and what duties have been imposed on the person causing the externality.

Valuing the tangibles

Production externalities for manufacturing and extractive industry, should be based on an estimate of lost output volume multiplied by the ratio of value added to output for the industry concerned. Data should be derived from ABS production data at the highest available dis-aggregation. Where the externality results in additional costs rather than lost production, it will be necessary to collect cost data from the affected establishments. For water-based commercial tourism (e.g. scenic boat trips), the valuation should be the number of days for which water quality conditions prevent operation multiplied by some percentage of the gross daily net revenue (our provisional best guess would be to make the fraction 50%, but this can vary with the enterprise and may require further investigation). A potential problem here concerns the diversion of tourist traffic to other operators or other days and times. Our advice assumes that the value is lost and not simply re-distributed. For delays to road traffic (probably mainly a consequence of stormwater flooding), the standard practice for transport improvement appraisal should be used. Estimated time delays x average wage rate for commercial traffic; estimated time delays x 30% of average wage rate for all other traffic including commuting.

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Property damage should be based on local land values where land is lost to erosion, or local property values where built structures are affected. This is likely to be a factor mainly for stormwater. Damage to infrastructure such as roads and railways should be valued at the costs of repair. Other property suffering stormwater damage, such as fittings and furnishing of houses, vehicles etc. should be valued at replacement cost. In valuing property damage particularly from sources other than stormwater, care should be taken to avoid double counting with production externalities. Any externality should only be valued once.

Fishery externalities. Should be valued at the difference between the carrying capacity of the receiving water at the current and target WQO (see below) multiplied by an assumed sustainable off-take percentage. For recreational fisheries, this should be multiplied by the unit market value of the species concerned. Species that are not marketed should be treated as wildlife. For commercial fisheries, the price weight should be 50% of unit landed value to allow for the costs of the fishery. This is a tentative judgement. Some studies suggest much higher costs (e.g. in some ocean fisheries). If the issue is important for water pricing, further investigation should be undertaken. Where impoundment has created a fishery that would otherwise not be there, it becomes a positive externality that should be a deduction from the price of the water service. Valuation should be on the basis of off-take with the other valuation rules as above.

Valuing partial intangibles

Recreation and amenity effects should only be considered where they are of major significance. Minor ones are incorporated in the treatment of wildlife and bio-diversity. Major effects might be the recreational benefits of impoundments (a deduction from the water service price) or the loss of opportunities from rivers and coastal waters polluted by sewage (an addition). For these cases, what is needed is an estimate of visitor days (number of visitors x average duration of visit) created or lost. This estimate might be weighted by the results of a contingent valuation or travel cost exercise designed to determine consumer valuation. In the absence of that, the conventional approach is to value at a fraction of the average wage rate. 30% is the standard valuation of leisure time in transport studies. A note of caution must be sounded here. There are alternatives to many such facilities so that the non-availability of one site might simply result in the use of another. In that case, the value of even major recreational and amenity losses and gains may be very low, amounting to no more than some additional time spent in travel, and arguably some reduced utility deriving from a less suitable substitute. For international tourists who, once in Australia, are a captive population for the duration of the stay, and whose destination decision is unlikely to be influenced significantly by conditions at a specific site, our view is that the externality should have zero valuation. A judgement is required as to the availability of recreational sites in the catchment, relative to demand. Only when there is perceived to be a supply deficit should any valuation of the externality be made.

Health effects. From what has been said above, the only issue to be considered briefly here is the valuation of injury and death from stormwater effects. Injury is typically valued at working days lost x mean wage rate plus medical costs. Deaths are valued similarly at working days lost through premature termination of life. This requires an estimate of the mean age of death through flood events. To this is added medical costs. Burial expenses are conventionally ignored although, in strict logic, some allowance could be made for the fact that these are incurred earlier than would otherwise have been the case. Finally, there is the element of pure intangibility. It is conventional to add (e.g. in the valuation of deaths through traffic accidents in road schemes) an arbitrary sum for 'grief, pain and suffering.' We make no recommendation on what this sum should be or whether it should be used. One possible means of estimation, used in the USA, is to look at the sums awarded by courts as compensation to the relatives in industrial accidents where liability is established. In UK road appraisal, the Ministry specifies a sum for use in cost benefit appraisal without explaining how this is derived.

Valuing Pure Intangibles

Wildlife and bio-diversity externalities should be calculated as the additional costs of achieving minimum target river water quality and flow level objectives which are derived from a duty of care. These costs will include the costs of upgrading sewage treatment works, improving stormwater control and treatment, and allocating or importing environmental water to the catchment, as is proposed for the Murray assuming that water is traded fairly. Imported water does **not** need to be valued at its opportunity cost, since this would involve double counting (the opportunity costs will already have been borne by those losing water rights). The charging base is the additional system costs for meeting water quality objectives. These costs are dependent on the COAG full cost rules. They should be calculated according to COAG rules for new investment, and include annual running costs, depreciation and required rate of return on the assets. When those costs are actually incurred and the environmental improvements are presumed to have been made, there should be no change in the price of the water service (provided that they have been correctly calculated and upgraded each year to current prices). In an accounting sense, the composition of the charge will have shifted from the payment for an externality, to payment for investment incurred.

It may be desired to bring about environmental improvements by changing behaviour, rather than making capital investments, and the tariff restructured to incorporate incentives to recycle grey water, separate grey from black water, etc. Incentive payments should be based on the savings in expenditure on the river water quality improvement programme that these behavioural changes lead to. The effects of more general incentives to save water (e.g. through multi part tariffs) are problematic because we lack knowledge about long-term price elasticities of demand.

Where wildlife and biodiversity objectives are poorly defined, the work may need to be underpinned by a series of studies to elucidate the nature of public preferences for wildlife and biodiversity conservation. These studies should then be used to propose a set of objectives or outcomes against which costs can be measured.

Putting it all together

Our proposals are summarised in Table 4 below.

Table 4 Summary of proposed methodology

Externality type	Valuation method
Production	Manufacturing and extraction: lost output x value added (ABS production data) Water based tourism: operating days lost x 50% of gross operating revenue (50% may be modified by investigation) Road traffic delays: time lost x average wage rate (commercial traffic) or 30% of average wage rate (all other traffic).
Property damage	Local land values where land is lost to erosion. Local property values or the cost of repair where built structures are affected. Costs of repair for roads and railways.
Fisheries	Difference between carrying capacity at current and target WQO x sustainable off-take x Market price (edible species in recreational fisheries) or 50% of landed price (commercial fisheries). Inedible species in recreational fisheries treated as wildlife.
Recreation	*Partially intangible.* Only major impacts valued, and then only if alternative facilities are in short supply. Where a valuation is thought necessary, estimated visitor days affected x either 30% of average wage rate or willingness to pay derived from a CV or travel cost study. Care should be taken to ensure that there is genuinely some lost value. Minor impacts are treated as wildlife and bio-diversity. *Pure intangible.* Full long-run marginal costs of upgrading from current to target water quality objectives.
Amenity	As for recreation
Health	*Tangibles.* Full long-run marginal cost of damage to property associated/caused by floods. *Partially intangible.* Injury and death (stormwater flooding) Estimated working days lost x average wage rate + medical costs. We make no recommendation on whether or not a sum should be added for `grief, pain and suffering'. *Pure intangible.* Full long-run marginal costs of upgrading from current to target water quality objectives.
Wildlife and bio-diversity	Full long-run marginal costs of upgrading from current to target minimum water quality objectives.
Resource management costs	Full long-run marginal cost to government of monitoring and managing externalities.

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Water quality standards and objectives

Throughout Australia, catchment management plans, water use licences and emission permits are used to define water quality objectives. Essentially, these arrangements set out the duty of care provisions noted earlier. To use the above valuation system for cost benefit analysis and/or to assess the appropriateness of water service prices, it is necessary for objectives to be presented as a set of indices. It is also necessary for dose-response functions to be built so that assessment can be made about the likely change in WQO when loads to the system change. We stress the importance of knowledge about dose response functions and the probability of their occurrence.

Water quality objectives can be defined in a number of ways. Some of the main ones are:

Classification by use. Criteria here could include:

- Potability - suitable for drinking water supplies with only primary treatment, needs secondary treatment, is unsuitable)
- Recreation - safe for immersion sports, suitable for non-immersion recreation, not suitable for recreation
- Fishing - well stocked with fish which are safe to eat, well stocked but unsafe to eat, poorly stocked, cannot support fish

As with the other measures potability, and recreation indicators would need to be specified by the number of days for which the WQO was reached since temporary failures (e.g. bans on use of the Torrens because of blue algae and of Sydney beaches from exclusion).

Biological status. Waters can be classified by the density and number of species of biota that they will support. This classification system utilises indicator species since total counts would be expensive to administer and subject to sampling hazard. The indicator species have to be relatively abundant so that lower order biota are indicated (perhaps including aquatic insects but not birds or mammals) and would have to include species whose density is sensitive to variations in levels of the main pollutants. One problem of this classificatory principle for a country as large and ecologically diverse as Australia, is that indicator species would have to vary . This makes comparison among catchments difficult.

Biological oxygen demand (BOD) and suspended solids (SS). These are the standard measures of sewage pollution. The problem with them as an index of water quality, is that there are many other chemicals that affect the environmental health of a water source. In Australia they would certainly need to be supplemented by salinity, although the sources of salinity are largely outside of the urban water system.

Chemical and biochemical composition. This classificatory system would extend the previous one to encompass other chemicals that are important to environmental health. This is probably the most sophisticated form of index and can be mapped to the other partial indicators.

WQO are commonly used for policy purposes in other countries. The UK has used them for policy purposes since the 1980's, and over time, the classificatory system has shifted from use measures through biological indicators to chemical indicators (Environment Agency, 1999).

Monitoring WQO should be based on regular sampling of waters. Standards would be based on the proportion of samples reaching the required standard (95% is the normal compliance criterion). For use indicators, such as immersion sports, the number of days on which the water is suitable is an alternative basis for classification.

With a set of WQO in place, intangibles are valued at the cost of meeting the WQO. Costs will include additional treatment of dirty water and measures to maintain flow levels.

Issues in the application of the framework

Multi-jurisdictional catchments

Ideally, the pricing of water services and choice of the mix of instruments to convey appropriate signals, including the valuation of externalities, is best carried out at the level of the catchment. This is particularly important in sequential systems, where the delivery of water services is divided between two or more water providers. The impact of investment by any one of them on WQO, will be conditional on the actions of the others, posing problems for setting prices. In these circumstances, WQO can only be jointly delivered through co-operative planning and action. One obvious and recommended strategy to achieve this, would be for all parties in the catchment to fund application of the methodology across the entire catchment. By these means, co-ordination savings could be anticipated and WQO delivered at least cost.

Small and large towns

The practical application of the pricing and valuation principles presented in this report requires the commitment of resources by the water industry. The tasks include:

- modelling dose–response functions;
- developing indices for water quality;
- putting in place mechanisms for monitoring performance against WQO; and,
- ultimately, formulating and ultimately implementing the investment programs needed for WQO.

These tasks may be beyond the resources of water service providers in small and even medium sized towns if acting alone. Thus, if progress is to be made co-operation is clearly required. If this does not happen, then the alternative scenario is one where the specification of WQO, and the programmes to upgrade, will be less comprehensive, and the pace of environmental improvement slower, in smaller communities. This may be unsustainable in the presence of interdependencies through sequentiality, as in the Murray-Darling Basin. Elsewhere, it is probably unacceptable on grounds of equity.

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The rural and the urban

Over 70% of total water use in Australia is for agriculture. A catchment will probably have interdependence between urban and rural water, use even if it is subject to none of the other problems discussed in this section. If we are to make progress with environmental improvement, it is necessary that the principles of pricing are broadly comparable between urban and rural water users. A methodology for valuing externalities associated with water use in agriculture is needed.

Summary and conclusions

This report is concerned with the methods by which externalities and resource management costs of urban water services should be valued, in order that they be incorporated into the full long-run cost pricing of water services and into benefit cost evaluations of managing the urban water system.

Economic efficiency requires that the externalities attributable to each water service be identified and charged to that service.

Water pricing may also be used additionally to achieve broader objectives of the urban water cycle and to encourage changes in user behaviour

Externalities produce social costs and benefits. Perception of these social costs and benefits is mediated through the evolving concept of a duty of care for the environment. While this notion is gaining acceptance among rural water users, it is new to urban water users.

Externalities can be classified in a number of ways. A critical distinction is between tangible externalities, where valuation can be derived directly or indirectly from prices prevailing in a market, and intangibles where no market exists.

With intangible externalities, the basic requirements for the existence of a market do not exist. Where these requirements are satisfied for either supply or demand, but not both, we have a partial intangible. Where they are satisfied for neither we have a pure intangible. The challenge of this report is to find means of valuing intangibles.

The key to valuing these pure intangibles (and some partial intangibles) lies in definitions of duty of care and WQO. The duty of care receives practical embodiment in WQO for each catchment. The value of intangibles is correctly measured by the cost of meeting WQO provided that WQO are seen as a requirement of catchment management policy.

After review, it is concluded that the techniques of contingent valuation, travel cost and surrogate markets, have a limited scope in providing money valuations for some partial intangibles, but are not practical and, arguably, not appropriate in principle, for pure intangibles. These techniques, however, can have value in elucidating the nature of public preferences for the delivery of intangibles.

The process of valuation has three stages:

- the identification and classification of externalities for each water service;
- determination of the physical scale of each externality through dose-response modelling;
- assessment of probabilities of occurrence; and the valuation of the physical quantities.

The classification of externalities has to be designed to avoid overlap and double counting. We suggest the use of a seven-fold classification of externalities.

Where the data, or the understanding of the dynamics of the water system, are not sufficient for dose-response modelling, a relationship may need to be assumed. Those interested in protecting the interests to the environment will argue that when making such assumptions, regard should be paid to the precautionary principle.

We provide a detailed discussion of the valuation methods for each class of externalities. The recommendations are summarised in Table 4.

Central to the valuation exercise is the specification of catchment WQO and the costing of an investment and management programme to achieve them. The costs of this programme should be in conformity with the COAG full-cost pricing principles. We discuss how these principles should be applied to the exercise.

WQO should be embodied in water quality indices. There are a number of alternative bases for constructing such indices that we review. It is important that performance under these indices be monitored at regular intervals and, for purposes of price setting, at least yearly. In a separate report (Young 1999), it is stressed the efficient externality management requires use of the full range of incentives instruments available. Effective full-cost pricing requires use of a mix of instruments.

Implementation of this framework will require co-operation and co-ordination in sequential catchments (where water is re-used), with the pricing of rural water, and among authorities in small and medium sized towns, where otherwise resource constraints compromise the full-cost pricing programme.

Recommendations for further work

Methodology for the valuation of externalities is politically sensitive and controversial. While basing our proposals on a firm grounding in the underlying economic principles, we have sought to be pragmatic and to push issues to the limit to achieve practical applicability. The next steps as we see it are as follows:

1. Testing the framework for a range of conditions – for small, large and medium catchments, for nodal and sequential systems and for catchments with both urban and rural water use.
2. Development of a set of water quality indicators to be applied across catchments as a critical step forward.

3. We have been surprised by the lack of information of the relationship between discharge loadings and water quality outcomes. We recommend that work on dose-response relationships be undertaken expeditiously.

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APPENDIX ONE DOSE-RESPONSE MODELLING

For an extensive summary of data available, readers are referred to the NSW EPA's ENVALUE system. This is available on the world wide web at http://www.epa.nsw.gov.au/envalue/

The starting point for any valuation of an externality is some sort of *dose-response model*. This models, in physical terms, the cause to the effect. Thus, if the problem is the emission of dirty water resulting in eutrophication of the water system, the dose-response model will predict the degree of eutrophication from the volume of dirty water discharges and whatever other variables affect the outcome. Dose-response modelling has to extend from the actions of the individual or body that is causing the externality, to the response of the individuals or bodies who perceive the external effect.

Initial cause ⇒ physical transmission ⇒ perception of effect
Control cost ⇒ dose-response model ⇒ valuation of externality

Thus, our example above would be appropriate if people found the eutrophication of watercourses to be visually offensive, i.e. if eutrophication was the externality. But it is more likely to be the effects of eutrophication that are of concern: e.g. the resultant modification of the ecosystem with changes in flora and fauna and hence in the degree of biodiversity. In this case, the dose-response modelling has to extend to the prediction of changes in an index of biodiversity. This means that a dose-response model could be quite complex, involving several sub-systems, requiring the collection of complex data and the input of several scientific specialists. Some observations are in order.

The transmission mechanisms may not be sufficiently understood to permit prediction of response from dose. Alternatively, the system model may require information which is not available and which cannot be provided, except at prohibitive cost. These problems frequently exist with impacts on wildlife and the natural environment. Thus, one may know in general terms that eutrophication reduces ecosystem diversity but have insufficient understanding of the process to quantify the relationship. Additionally, the measurement of ecosystem diversity is itself far from straightforward.

Where effects have persisted over long time periods, there is a problem of the initial point of reference. It is widely agreed that impoundment and abstraction from the Murray River has had profound impacts on its estuarial ecosystems. The first impoundment occurred last century and abstraction has increased with further impoundments from that date. From which date (and hence from which flow rate) does one measure the externality?

This problem is avoided in the cost-benefit analysis of alterations to the system (as might be proposed in a catchment management plan) since the status quo ante is then the point of departure. Thus, the problem of assessing a new impoundment on the Murray may be tractable, although there is still a need for a dose-response model. In the simple textbook notion of the optimum state of the environment, the problems of measuring total and incremental impacts are avoided by the assumption of a fully known (and probably linear) dose-response function. In such a world, any pair of dose and response levels can be chosen by reference to control costs and valuations of the externality. In practice, such functions often require considerable modification before they even begin to describe reality. Unfortunately, such data is not always available.

Where it is impossible to specify the precise dose-response relationship, but there is confidence that one exists (e.g. that reduced water flows reduce ecosystem diversity), there may be a case for assuming the function. Where necessary, expert judgements can be used to guesstimate the most likely form and magnitude of the response associated with a "dose". The assumption will probably be of a linear mapping from cause to effect and derived through consultation with experts. Where the impacts on the water system are thought to be irreversible, as may be the case where endangered biota are found in the catchment, proceeding on an assumed dose response relationship may be justified by reference to the precautionary principle. Of course, if the assumption is wrong, the outcome may not be what was intended.

APPENDIX TWO TECHNIQUES FOR THE MONETARY VALUATION OF EXTERNALITES

Given predictions of responses from the dose-response model, there are broadly four main techniques for converting them into monetary values:

- Market valuation;
- Revealed preference techniques;
- Stated preference techniques;
- Neutralisation cost.

We treat these in turn. In each case, we state the principles before illustrating with examples.

Market valuation

Market valuation techniques are only possible where actual markets for the effects exist. This can only be the case for tangible externalities. Where a market valuation exists, it should always be used in preference to any of the other techniques discussed below. Thus:

- ➢ where possible, tangible externalities should always be given market valuations.

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In the presence of a tangible externality, there will, in principle, exist two market valuations: the recipient's valuation and the provider's valuation. These will differ because the market has not 'cleared'. Choice should be based on the least cost way of resolving the problem. This will depend among other things on the numbers of providers and recipients. As an example, consider a situation where discharges from an urban sewage outfall reduce water quality for a manufacturing establishment located downstream, forcing it to treat its water or to purchase clean water from another catchment. The two market valuations are:

- the factory's valuation of clean water, measured by the delivered cost of water from the other catchment (unit price x volume purchased); and
- what it would cost to reduce the nutrient loading of the sewage discharges to a level where the factory could use the water (engineering estimates of the costs of the necessary treatment).

Where alternative market valuations exist, choice between them should be made on the basis of the least cost method of eliminating the externality.

Revealed preference techniques

While no direct market value exists for them, it is possible to make some inference about amenity and recreational values from observations of expenditure on other goods and services. There are two basic techniques available:

Travel cost technique. Some idea of consumer valuation of amenities and recreational resources such as national parks, viewpoints and beaches may be inferred from the money that they are prepared to spend in getting to and using them. The notion here can be illustrated by a simple example. If an individual or a family is prepared to spend, say, ten dollars in petrol and parking charges on visiting a national park we may infer that they place a value of at least ten dollars on the resulting recreational experience. This simple idea can be elaborated to yield an estimate of the recreational and amenity value of the facility. The methodology requires a sample survey of visitors to provide the information on travel expenditure (mode, time and distance) demographic characteristics (age, sex, family size) and purpose of visit.

Surrogate market technique. While people do not directly buy amenity as a separate good or service, they may buy it indirectly as a quality or characteristic of something else. Where this is so it is possible, in principle at least, to isolate the value of amenity. Almost all the work under this technique relates to property prices. It is observed that property prices are affected by visual amenity. Properties with views of beaches, national parks etc. command a premium. Equally, proximity to facilities of this type can also raise prices. The property price premium is viewed as the capitalised valuation of the amenity and studies seek to isolate the role of amenity from the multitude of other factors affecting the price of property.

Recreation and amenity are partial intangibles. Revealed preference techniques have been widely used for valuing them, particularly in the context of cost benefit studies. There are many examples of applications to the water system including environmental enhancement.[6] The techniques can be viewed as substitutes, with choice between them determined by the nature of the problem. Even so both

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6 Bicknell (1998) has recently used of house price analysis to measure the amenity benefits of planting native vegetation on urban watercourses in Wellington NZ.

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techniques pose difficulties to the analyst. The travel cost technique requires that site visits are the intended and sole purpose of visitor journeys. This is often not the case. Isolating the amenity component of house prices requires statistical control for the other factors. In practice, the set of price determinants cannot be fully specified, some components cannot be accurately measured, and collinearity between determinants is a problem. In consequence, estimates of valuations for similar phenomena can vary widely between studies and between the two techniques. It is not possible from the available literature, to derive anything like standard valuations, so any use would require a special study. None-the-less, the use of one or other should be considered where changes in the catchment water system, or major developments in stormwater management, are expected to have substantial recreational or amenity impacts. In these circumstances contingent valuation (see below) is an alternative.

Stated preference techniques

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Stated preference techniques rely on questionnaires administered, ideally in face to face interviews, to a statistical sample of those who are experiencing the externality. The intention is to determine what respondents would be willing to pay in a hypothetical market for the externality. Hence, these techniques are alternatively known as hypothetical market techniques. The major hypothetical market technique is contingent valuation (CV). CV might be described as the flavour of the decade in environmental economics. It has been used to impute money values to a very wide range of environmental externalities and has given rise to a large amount of academic literature.

The essentials of CV are simple. Respondents are given a description of an environmental problem, a proposed solution, and the means of financing the solution (the payment vehicle). They are then asked what they are willing to pay for correcting the problem. Willingness to pay is usually achieved by some form of bidding process initiated by the interviewer. As an example, the problem might be pollution of water at a bathing beach through the discharge of sewage. The proposed solution is to upgrade the sewage treatment works to tertiary treatment levels. This will be financed by an increase in the unit price of metered water. Respondents can be asked to state the increase in unit price that they are willing to accept. Alternatively, and preferably, the interviewer will ask if they would be willing to see price rise by, say, and 10c. per litre. If this is accepted then the suggested price rise is increased to say 15c. If it is rejected the price rise is reduced to 5c. The bidding process is iterated to determine the maximum willingness to pay.

The attractions of the technique are obvious. It is easy to understand and is flexible. It can be applied to almost any externality, to tangibles and to both partial and total intangibles. None-the–less, there are problems with it.

Response biases. A large number of potential biases are recognised in the literature which can be minimised, but by no means eliminated, by careful research design. Where the externality under investigation is an intangible, the extent to which biases are distorting the results cannot be determined by reference to alternative measurement techniques.

Treatment of outliers and zero bids. The normal practice is to take an average bid (usually the arithmetic mean or the median; sometimes the geometric mean) elicited from the questionnaire, as the basis for estimating the valuation of the externality. However, a CV study typically yields a substantial number of respondents who will bid nothing and a (usually smaller) number whose bids are wildly in excess (perhaps a hundred times) of the average. The conventional practice is to exclude the high bids as outliers. Zero bids are sometimes excluded but more typically now included in calculating the average. How zero bids and outliers are treated has serious implications for the calculation of the average. The rationale for exclusion of the outliers is that the respondents producing these bids are failing to understand the nature of the exercise. This might equally be said of those who bid nothing and indeed of those whose bid is close to the average.

Aggregation. Once the average bid has been determined the value of the externality is calculated by grossing it up to the population of which the bidders are assumed to be a random sample. But how is the population to be defined? It should be those who experience (suffer from, or for a positive externality, benefit from) the externality. This population is unlikely to be coterminous with the population at risk of paying for the correction of the externality. Thus, in our example of the pollution of the bathing beach, users may include tourists from overseas or other states or intra-state visitors from other catchments. With a partial intangible such as recreation or amenity, the population can probably be defined, but for a pure intangible, such as wildlife, it probably cannot. Are the sufferers from the destruction of Australian wildlife the population of the affected state, or the nation, or given Australia's international commitments to conserve bio-diversity, the population of the whole world? While choices about the means of averaging responses can make a difference of a few cents, choices about the population can alter the valuation by millions of dollars. This issue is unresolved in the CV literature and examples exist of all alternatives.

Replicability and consistency. If CV exercises are meaningful, then they should be capable of replication with repeat studies yielding results that are broadly comparable with the originals. Equally studies should be consistent in other ways. Thus, a study eliciting bids to conserve one example of a threatened habitat, should produce lower valuations than proposals to conserve many or all examples. It has been shown (Meade, 1993) that CV exercises fail one both counts.

Feasibility. CV is a feasible technique for the estimation of tangible and partial intangible externalities such as recreation and amenity where respondents have some understanding of the problem and some basis for expressing a willingness to pay. It faces serious difficulties with total intangibles such as wildlife and bio-diversity. Bowers (1997) identifies two major and over-whelming problems:

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Impossibility of accurate description. Elicitation of an accurate bid depends on a complete and accurate description of the problem and its solution. With impacts on the natural environment, this is not possible in the time allocated to each interview. While it is believed in general terms that, e.g. increases in dry season flows or reductions in nutrient loadings of rivers, will lead to benefits in biological diversity, the precise effects and the importance for the overall objective is not fully understood. Hence, the understanding cannot be imparted to bidders. In those circumstances, respondents seize on to the things that they do understand such as increases in the price of water and react to those;

Lack of competence. CV is asking respondents to use their experience in operating in real markets, to place values on trading in hypothetical markets. But they have no relevant experience to bring to bear. Consumers learn what are appropriate values in the markets in which they operate: on what is a reasonable price for a kilo of oranges or a property in Adelaide. They learn in a number of ways: by shopping around; by talking to others; reading the press etc. None of this helps them in hypothetical markets for wildlife, bio-diversity, or atmospheric integrity. Furthermore, the methods of acquiring an understanding of appropriate values do not exist since the market is no more than hypothetical. In the circumstance, it is not surprising that some refuse to bid and others offer large sums. The bids that they make are no more than artefacts of the research process. They have no external validity and are not an appropriate basis for taking policy decisions.

In sum, CV is a possible technique for assessing tangible and partial intangible externalities. It does not yield reliable money valuations of pure intangibles. At best, it gives a ranking of people's perceptions of preferences if asked to vote on an issue. This can be invaluable in elucidating the nature of public preferences for delivery of intangibles. Having made this judgement, however, we admit that others are of a different view. It can be argued that non-market valuation of pure intangibles is valid. Nevertheless, it is our perception that the neutralisation cost technique described below is easier to both explain and defend. It is based on standards developed through the political processes associated with the development of catchment management plans and legislation.

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In situations where the intention is to transfer the results from a contingent valuation study from one location to another, as a general rule choice modelling rather than contingent valuation should be used. Choice modelling collect data on the exogenous factors that explain each the values collected. This increases the validity of transferring results from one area and one situation to another.

Neutralisation cost

The cost of correcting an externality is, in normal circumstances, no measure of its value. Bowers (1997) shows this with an example of a road-widening scheme that entails the demolition of a bridge carrying a railway line. The externality is the loss of the railway route. The value of that loss depends on the volume of traffic that the line carries. If it is disused there is no loss. On the other hand, if it is a major passenger or freight route the loss could be very large. The cost of replacing the bridge and correcting the externality, however, is the same in both cases.

The cost of correcting the externality becomes relevant if avoiding the externality is a constraint upon decision-making, since then the expenditure must actually be incurred. Thus, if it were a requirement, embodied in regulation or legislation, that improvements to the road network should not reduce the availability of other transport infrastructure, replacing the bridge would be incorporated into scheme design and appear as part of scheme costs. There would then be no need to measure the value of the rail route in a cost benefit analysis of the road-widening scheme since the scheme would not change that value.

Engineers have long recognised, and worked with, constraints on scheme design. Important ones are safety and health constraints. Infrastructure works, such as bridges, incorporate a safety margin above maximum expected loading in their design. Similarly, cambers on road bends are designed to provide a safety margin above maximum recommended speeds. In sewage treatment works, there are similar health and safety constraints. In designing structures, the engineer does not treat the safety level as a variable to be subjected to a cost-benefit test, reducing standards if economics dictates. A minimum standard is a requirement dictated not simply by law and regulation and enforced by legal sanctions, but embodied in professional ethics. Imposed constraints are equally not dependent on customer approval. If a road bridge is built to connect an island community to the mainland financed in whole or part by the island residents, they cannot ask for reduced safety standards so as to reduce the charge.

Constraints, such as health and safety, are derived from general *duties of care*. A duty of care exists for the public safety and similarly for public health. A duty of care, as imposed upon engineers and other public servants, contains three conditions:

- a duty to meet at least minimum standards;
- a duty to meet those standards in a cost-effective manner; and
- a duty to raise those standards as technology and finance permit.

The basic way to deal with intangible effects of the water system is through the duty of care for the environment (Young, 1999). This duty of care has evolved through public concern and legislation reinforced by international, national, state and local commitments to sustainable development and conservation of bio-diversity. This duty of care for the environment, relates to risks of irreversible damage through destruction of wildlife and ecosystems. While similar to other duties of care, it is different in that there is widespread agreement that the current level of performance is unsatisfactory. Thus, the duty of care for the environment as applied to the water cycle requires that environmental quality be increased. The duty of care may thus be summarised as:

- a duty to maintain and enhance the quality of the water environment to the benefit of wildlife and other users; and
- to do so in a cost-effective manner.

This duty of care for the water environment is often operationalised in WQO in most catchment management plans and similar documents.

APPENDIX THREE PUBLIC GOODS

Public goods lack one or both of the essential characteristics needed for the operation of a market. These characteristics are:

Excludability Property rights enable a seller to exclude non-buyers from consuming the good. Where a good is excludable, the seller transfers a property right to the buyer.

Rivalness A market transaction involves sacrifices by both parties. The seller is deprived of the services of what he sells and the buyer sacrifices opportunities to devote his resources to something else.

These sacrifices are termed opportunity costs in economics and in a market economy are measured by prices. The fundamental condition for the operation of a market is as follows:

> *For a transaction to take place, opportunity costs must be positive for both parties, and the opportunity cost to the buyer must be equal to or greater than the opportunity cost to the seller*. [When the market clears (is in equilibrium) opportunity costs at the margin for buyers and sellers are the same].

If a good lacks excludability then it can be consumed without purchase. The opportunity cost to the buyer in this case is zero and the fundamental condition cannot be satisfied.

If a good lacks rivalness then the opportunity cost to the seller is zero and again the fundamental condition cannot be satisfied.

The argument is summarised in the Box A3.1 below:

Box A3.1

Market good or service. Opportunity cost equals market price and is the same for both buyers and sellers *Tangible externality.* No market exists but a market is feasible. Because there is no market either too much (negative externality) or too little (positive externality) is supplied. Opportunity cost could be measured from either side of the potential market but it is normally measured from the short side i.e. the sufferer's valuation for a negative externality and the provider's valuation for a positive externality. *Partial intangible externality.* A market cannot exist because the opportunity cost is zero for one side (good or service is non-rival or non-excludable but not both). It might in principle be possible to measure the positive opportunity cost of the other party and some of the methods discussed in this report attempt to do that. *Pure intangible externality.* A market cannot exist because opportunity cost is zero for both sides. (The good or service is both non-rival and non-excludable). There is no opportunity cost valuation.

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