CRC for
Water Sensitive Cities

Nonmarket valuation of water sensitive cities: current knowledge and issues

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Australian Government
Department of Industry, Innovation and Science

Business
Cooperative Research Centres Programme

Nonmarket Valuation of Water Sensitive Cities: Current Knowledge and Issues
Valuation of economic, social and ecological costs and benefits (A1.2)

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Publisher

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Date of publication: December 2016

An appropriate citation for this document is:

Zhang, F., Fogarty, J. (2016) *Nonmarket Valuation of Water Sensitive Cities: Current Knowledge and Issues.* Melbourne, Australia: Cooperative Research Centre for Water Sensitive Cities

Contents

Abstract

This paper presents a systematic review of the application of the economic evaluation methods that are relevant to Water Sensitive Urban Design (WSUD). WSUD involves integrating the urban water cycle into urban design to improve water supply and environmental protection. The review considers four main WSUD-related aspects: improving and securing water supply requirements; protection of groundwater systems; management of wastewater; and environmental protection. The literature reviewed is grouped under these broad headings, and the evaluation method used to obtain information about non-market values. The advantages, disadvantages, and limitations of each non-market valuation method are also summarised and compared.

The review establishes that the two methods most commonly used to estimate non-market values for benefits relevant to WSUD have been contingent valuation and choice experiments (also known as choice modelling). Other valuation methods, such as the travel cost method, the averting behaviour method, the hedonic price method, and engineering methods have also been used.

For some areas of benefit that can be delivered through WSUD there is a reasonable knowledge base; yet in other areas the knowledge base is quite limited. The most appropriate way to generalise non-market valuation study results from one location to others remains unclear and is an area requiring additional research.

Introduction

Water Sensitive Urban Design (WSUD) is a land planning and engineering design approach that integrates the urban water cycle — including water supply, stormwater, groundwater, and wastewater management — into urban design.[1] WSUD can provide benefits that are easily quantified, such as additional water supply (Brown and Farrelly, 2009); and benefits that are not easily quantified, such as mitigating environmental degradation, improved aesthetic appeal, and the provision of recreational benefits (Morison and Brown, 2011).

An important element of the urban water management system is to provide sufficient clean water to both residents and the environment (Mitchell, 2006). Some urban water supply systems are struggling to meet these twin objectives. There are many reasons behind the current stress on urban water supply systems. In the developing world, population growth and rapid urbanisation are two factors impacting effective operation of the water-supply system. In the developed world, stress on the water supply system may be the result of: population growth, changes in rainfall patterns, poor infrastructure investment decisions, inappropriate historical water allocation decisions, and changes in the population's expectations regarding water management.

Aspects of WSUD, such as harvesting stormwater for future use and recycling wastewater for reuse, can assist with meeting the urban water supply requirement (Wong, 2006). Received stormwater can be stored in tanks, surface storage areas (such as lakes, waterways, and constructed wetlands) or underground. Stormwater harvesting can increase water availability in urban areas and by reducing stormwater runoff it can also help prevent the degradation of ecosystems (Roy et al., 2008). The environmental benefits from managing stormwater runoff arise because large-volume stormwater flow events are a major source of pollution in urban areas.

More generally, wastewater reuse is now recognized as an important element in addressing the global water scarcity problem (Wintgens et al., 2005). Treated wastewater can be used in a wide range of applications, such as agricultural or industrial applications, and can also be used to deliver environmental benefits (Bixio et al., 2006; Dillon, 2000). In the Australian context a specific environmental application of interest for recycled water is its use to recharge aquifers to maintain underground water levels and mitigate the effect of groundwater over-extraction (Mills, 2002).

At any given physical site there are various WSUD technologies and practices that may be appropriate. Economic analysis can help to evaluate the relative performance of these different technologies and practices, in terms of value for money. Here we make the distinction between economic analysis, which considers wider community/society benefits and financial analysis, which is used to measure the difference between project expenditure and revenue for an individual business. In other words, the external effects that are ignored in financial analysis are captured in economic analysis. The aim of economic analysis is to determine whether the overall welfare of the community/ society as a whole will increase from the proposed project.

In the economics literature, formal definitions of an externality are well established (Buchanan and Stubblebine, 1962). As a practical matter, an externality arises where the legal, legitimate actions of one party impact on the welfare of other parties external to any market transaction. Externalities can be positive, in the sense that they provide a benefit to the other party; or negative, in the sense that they impose a cost on the other party. Externalities can also be directional or reciprocal. In the context of water infrastructure projects, the externalities that arise might include: recreational benefits from water-quality improvements attributable to the use of modern wastewater treatment plants; ecosystem benefits from aquifer recharge; urban pollution mitigation due to advanced stormwater management systems; and ecosystem loss due to dewatering activity associated with a mining project.

[1] www.watersensitivecities.org.au [accessed 11 September 2013].

Figure 1 below provides an illustration of how externalities result in the quantity of a good consumed deviating from the socially optimal level of consumption such that there is a welfare loss with unpriced externalities. In **Figure 1**, the plot on the left shows the case where the marginal social cost of production is greater than the marginal private cost of production due to an externality. The socially optimal point of production is the point Q*, but the actual point of production is the point Q'. In the plot, the grey shaded area has an interpretation as a measure of the welfare loss associated with failing to take into account the negative externalities associated with production. The plot on the right in Figure 1, illustrates the case where there are spillover benefits that accrue to society in addition to the private benefits that accrue to individuals. In the figure the actual production level and the socially optimal level of production are again identified as the points Q' and Q*, respectively. In this scenario there is too little production, and again the grey shaded area can be interpreted as a measure of the societal welfare loss associated with under production of the good or service.

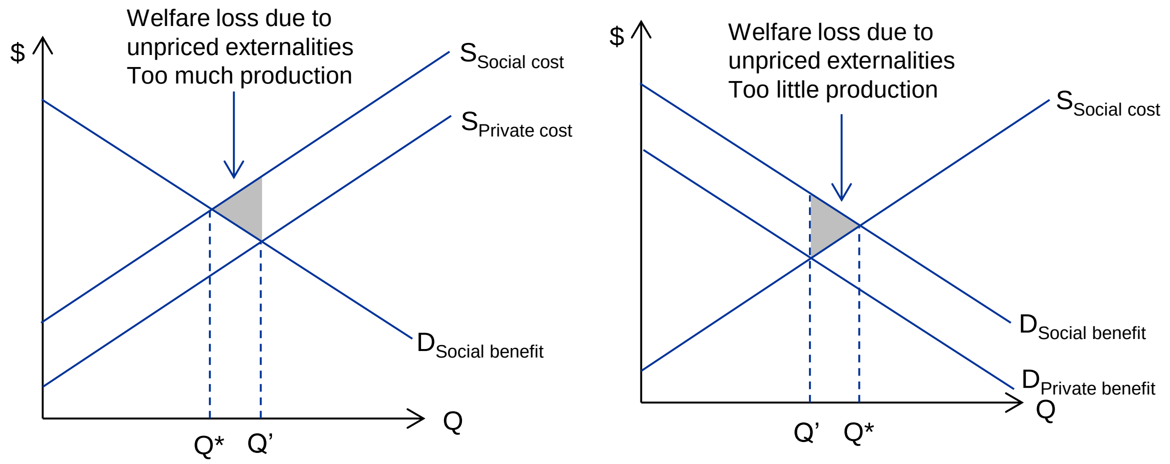

Figure 1: Illustration of the impact of externalities

Over the past decade water utilities in Australia have made substantial investments in a range of different technologies to augment water supply to urban areas. These investments have included: dam expansion projects, such as the Hinze dam expansion plant in Queensland and the Cotter dam expansion in the ACT; construction of desalination plants, such as those built in Western Australia, Victoria, and New South Wales; water recycling projects, such as the western corridor recycled water projects in Queensland and the Alkimos wastewater treatment plant in Western Australia, and various small scale stormwater harvesting projects. In total, the capital investment in water augmentation projects over the period 2005/06 to 2011/12 by Australia's largest water utilities is thought to have been around $30 billion (Productivity Commission 2011). The scale of infrastructure investment in the water sector is therefore substantial.

It is not necessarily the case that a water-conserving project will stack up economically. For example, in net present value terms, the most robust estimates available suggest that over a 20-year period the expected welfare loss to the Victorian community from the construction of a large desalination plant, relative to alternative lower-cost options of managing water supply, is between $2.7 and $3.7 billion (Productivity Commission 2011). Water infrastructure projects should be evaluated against economic criteria and shown to be economically viable once all the social and environmental considerations have been considered. At the moment, this is not the case. Although the appropriate framework for project evaluation is understood, there are practical difficulties regarding the estimation of the social, economic, and environmental impacts of projects required for a complete economic evaluation.

In most applications the market price for a good or service would be a basic building block in the economic evaluation process. The market price provides clear information on the extent of private benefits to purchasers of a good. The social and environmental costs and benefits would then be used to augment this initial market-derived value. However, in the case of water markets it is often the case that there are government supply subsidies, and or restrictions on where water can be sourced from. This in turn means that even the market price can be an unreliable indicator of value.

Additionally, the non-market valuation methods normally used by economists to capture the monetary value of environmental goods and services have limitations, and are not universally applicable. Although there are several different conceptual approaches, the two main groups of non-market valuation methods are revealed preference methods, which include the travel cost method and the hedonic price method; and stated preference methods, which include the contingent valuation method and choice experiments. The main difference between revealed preference methods and stated preference methods is that the former estimates the value of environmental goods and services based on observed real-world consumer behaviour, while the latter relies on information from community surveys in which respondents are asked about hypothetical scenarios.

The main limitation of the revealed preference method is that, as it is based on observed consumer behaviour, the approach can only capture information on the “use values” associated with assets. Use values are the benefits from direct or indirect utilisation of natural resources. Non-use values are benefits that accrue from environmental resources without a person directly using them. Non-use benefits include option value, existence value, and bequest value; and none of these benefits are captured in revealed preference analysis. Both use and non-use values can be estimated using stated preference methods, although stated preference methods in turn have a range of limitations. These include problems with survey respondents not having enough information to understand the nature of the trade-offs they are being asked to make, and general issues regarding the validity of values inferred from hypothetical scenarios where real money transactions do not take place (Nunes and van den Bergh, 2001).

In addition to the main stated preference and revealed preference valuation methods there are a number of other methods that can be used to obtain information on non-market values. These additional approaches include: the averting behaviour method, which is based on cost analysis; and the dose response method, which is based on examining the physical process of environmental impacts and estimating the losses (or avoided losses) from environmental degradation (or environmental quality improvement). The focus on costs, or avoided costs, distinguishes these methods from the revealed preference and stated preference methods that focus on benefits.

A major issue with all non-market valuation methods is that studies almost invariably relate to a specific site at a specific point in time. Values obtained from one specific site, using one specific valuation method, are generally not transferable to another context (Boyle and Bergstrom, 1992; Morrison et al., 2002). Yet because non-market valuation studies are expensive and time consuming to complete, there is a strong temptation to apply values obtained from one case study to other contexts.

The difficulties of undertaking comprehensive economic valuations means that, despite there being a range of different WSUD technologies installed around the world, the extent of comprehensive economic evaluations of WSUD is quite limited. The WSUD-specific evaluations that have taken place to date have been relatively simple, and have mostly relied on rules of thumb to infer benefits, or have discussed benefits in a qualitative manner only (Royal Haskoning, 2012; Gordon-Walker et al., 2007; USEPA, 2007; Roseen et al., 2011). As a first step in bringing together the knowledge required to undertake economic evaluations of WSUD projects, here, the general literature on potential benefits associated with water investments are summarised. The literature summary is structured as follows. First, for each aspect under consideration, a narrative summary of the existing findings is presented. A summary of the literature is then presented that outlines key messages. The final aspect of the literature review is a summary table. A key feature of the summary table is that study values reviewed have been converted into a common metric: 2012 $US. The summary table allows the reader to quickly gain an overview of the literature.

Valuation methods

The methods used to estimate benefits in the water economics literature have been: the averting behaviour approach, contingent valuation, choice experiments, hedonic pricing, the travel cost method, the cost of illness method, the stage damage method, and the photo projective method. A brief overview of each method is presented below.

Averting behaviour approach

The averting behaviour or averting cost approach estimates values through examining the costs that consumers incur if a service is not available. For example, if the quality of tap water is not at the drinking level standard, averting behaviour would include purchasing bottled water, installing purification devices in the home and office, and the regular boiling of tap water. If tap water was raised to drinking standard, the value of these activities would represent the costs averted by increasing the quality of tap water to drinking standard. Consumers may, however, have been willing to pay an amount substantially greater than this for the convenience of having drinking quality water available in the home. The averting behaviour approach can therefore be seen as finding the lower bound estimate to consumers' willingness to pay (WTP) for the improvement of environmental goods and services.

Stated preference techniques

The contingent valuation method relies on creating hypothetical market scenarios, and is a specific type of stated preference technique. The contingent valuation method seeks to uncover individual preferences for changes in the quantity or quality of a non-market good or service in the format of individual's willingness to pay. Using this method respondents' WTP for an environmental good is asked directly, and historically the contingent valuation method has been the most commonly used stated preference method in environmental economics research (Carson et al., 2001). An example of a representative question format typical of the contingent valuation approach is as follows: *Would you pay $X every year, through a tax surcharge, to support a program to improve water supply services?* An advantage of the contingent valuation method is that it can capture the public's reaction to each pricing level and establish an upper bound estimate of the value of changes in environmental conditions. This upper bound value can then be used by policy makers when considering investment decisions (Wang et al., 2010).

A common criticism of the contingent valuation method is that the method may not be able to capture the true value of an environmental good or service because people may not answer truthfully. Respondents may intentionally understate their true value or seek to 'free ride' on the responses of others, which in turn leads to invalid results (Lindsey and Knaap, 1999). It is argued that the choice experiments approach can overcome this problem because respondents are asked to choose among alternatives, and that represents a more realistic decision framework (Alberini and Kahn, 2006). For this reason, choice experiments are increasingly seen as preferable to contingent valuation for most environmental asset valuation applications. The other common criticism of the contingent valuation method is that the value derived from this method is sensitive to the level and extent of information provided by the respondents (Wang et al., 2010).

Choice experiments, as applied to nonmarket valuation scenarios, is a technique that comes from the conjoint analysis literature of marketing. In marketing applications conjoint analysis is used to determine the attributes of goods that consumers see as important. In environmental economics applications choice experiments may be thought of as a generalisation of the contingent valuation method (Snowball et al., 2008). With choice experiments, consumers are not asked directly how much they would be willing to pay to achieve some specific environmental improvement. Rather, respondents are asked to choose their preference from a series of alternatives which differ in terms of the attributes and the levels of attributes (Bateman et al., 2002). One representative choice experiments question is as follows: *Which one of the following schemes do you favour and which one would you be least likely to choose? Please keep your financial conditions in mind while answering.*

Note that one of the options presented to respondents is the below example of a choice sets (as shown by Table 1). A status quo option that allows the respondents to select the option of no change in environmental conditions at no cost is a feature of all choice sets.

Table 1: Illustrative example of a choice set of the attributes and levels of customer water supply services

	Option 1 (Status Quo)	Option 2	Option 3
Without warning your house might be without water from …	5:30am to 11:30am	5:30am to 9:30am	5:30am to 7:30am
In the last year, your water supply has never been interrupted. The water supply company tells you that your water supply might fail…	Two more times in the next 12 months	One more time in the next 12 months	No more times in the next 12 months
You are advised about the interruption by…	A card put in your letter box after the interruption	A phone call to let you know what was happening	A knock on your door by a company representative
The alternative water supply arrangements offered were…	None unless you requested it	Water was provided at a central location (water tanker in the street)	A 2 litre bottle of water was delivered to every household where someone was at home
As part of the package your annual water bill will…	Stay the same	Increase by $40	Increase by $80

Source: MacDonald et al. (2005)

Both the choice experiments method and the contingent valuation method rely on survey techniques and have specific strengths and weakness. An advantage common to both techniques is that they involve public opinion in the decision making process. Both methods also allow use and non-use values to be estimated which is a clear advantage of these methods (Bennett and Blamey, 2001). The main difference between these two methods is that choice experiments allow the valuation of the characteristics or attributes of the environmental good or service whereas the contingent valuation method arrives at an estimate of the environmental good or service as a whole (Bateman et al., 2002).

One criticism of the choice experiments method is that it assumes respondents view the sum of the attributes as equal to the whole value of an environmental good or service, which may be an invalid implicit assumption (Louviere et al., 2000). Using the choice experiments method, respondents are also required to understand the differences in each option where multiple attribute levels are varied. The relative complexity of the question format means that there are concerns about respondents' using decision heuristics to simplify their decision-making process. If respondents do fall back on simple decision heuristics when responding to the questions in a choice experiment survey, the results from the study are biased. A detailed discussion of this issue is presented in Bennett and Blamey (2001).

Revealed preference techniques

The basic premise of the hedonic price method is that the price of a market good is related to its characteristics, or the services it provides. This method is most commonly applied to estimate the value of local environmental attributes through modelling the variation in house prices. The central idea is that the value of a house can be decomposed into a set of main characteristics, such as size of lot, building area, number of bedrooms, or distance to the city centre; and social and environmental characteristics such as the crime rate, whether there are schools and universities nearby, proximity to environmental assets such as wetlands, etc. The hedonic regression approach treats the hedonic good as weakly separable in the consumer utility function such that consistent estimates of an implicit price for each attribute can be obtained.

There are generally accepted standards available for property valuations, such as Uniform Standards of Professional Appraisal Practice (USPAP) in the USA; Generally Accepted Valuation Principles (GAVP) in Germany; and Australian Property Institute (API) Valuation Standards in Australia. These standards help establish acceptable general equations considering different characteristics. Another advantage of the method is that the required house price data are generally available in a relatively open and transparent market. Thus, although the statistical issues involved in the estimation of a hedonic price model can be significant, the method is often the least difficult to implement.

The travel cost method is especially popular for estimating recreational values (Ward and Beal, 2000). It aims to convert the physical and social benefits produced by outdoor recreation, such as river, dam, and beach visits into monetary terms (Ward and Beal, 2000). The basic theory behind the travel cost method in valuing non-market goods, especially recreational sites and recreational activities, is that the travel cost is the implicit price visitors pay for their trip to access sites or to be able to take part in particular activities (Becker et al., 2005; Phaneuf and Smith, 2005). Through analysing the relationship between the travel costs (price) in accessing a recreational site and the number of visits per year to this site (demand), a demand curve relating the two can be found. An advantage of the travel cost method is the consistency with consumer demand theory, that is, the higher the cost, the fewer the visits. One major limitation of this method is that non-users are normally not sampled, therefore only use value can be captured (Ward and Beal, 2000).

Other methods

The other methods that have cited in this review include the cost of illness method, the stage damage method and the photo projective method. The cost of illness method has been used to evaluate the economic benefits of reduced illness from water pollution by estimating the direct medical costs associated with an illness (Van Houtven et al., 2008). The stage damage method has been used to estimate flood damage based on the understanding of physical processes of flooding (Smith, 1994). The photo projective method has been used to estimate the aesthetic value of water through asking people’s perceptions using photographs.

Water supply and pricing

One of the most important tasks for successful water management is to provide adequate and good-quality water to the public, at a reasonable price. In this section, we will discuss the water pricing mechanism, customers' responses to possible water price changes, and the economic value of a high quality water supply to the public.

Water supply infrastructure

Water for direct use has a market price, but as a public good, the market price does not reflect water's real value. Governments and the public have an overwhelming influence on the water price even if the main water suppliers are operated as a for-profit venture (American Water Works Association, 2000). Although, most of the water supply utilities are either publicly owned or operated as not-for-profit, competition in the local provision of retail water is normally not viable due to the great expenses of replicating water delivery infrastructure. For business such as water suppliers, fixed costs are high. The marginal cost to serve one more customer is, however, normally small and constant. Water supply companies are therefore natural monopoly companies. For a certain water demand level, the long-run marginal cost of the natural monopolist is lower than the long-run average cost. For these companies to be able to cover their costs, without government subsidies, the price of water should therefore be set to long-run average costs. This natural monopoly feature is recognised in Australia, where water supply companies are typically regulated by government.

Note, however, that as a scarce resource, the short-run marginal cost of water could increase significantly after a certain supply level (Bhattacharyya et al. 1995). This is because when low cost raw water resource are no longer available, the water utilities have to use more costly options such as seawater desalination or long distance pipe lines for additional water supplies (Howe et al., 1994). In this case, to cover the cost, the water pricing should not be set below short-run marginal costs (Shaw, 2005).

Water pricing and demand

In the economics literature it has become common to report the response of consumers to price changes using the elasticity metric, which is a unit free measure. In addition to being a unit free measure, price elasticities, which in general terms measure the change in quantity when price changes, can be derived from the generalised demand equations of consumer theory (Gravelle and Rees, 1992). Consumer theory is well developed, and as such, consumer theory can be used to gain an *a priori* understanding of the factors that impact the water own-price elasticity.

Formally, the own-price elasticity of demand for water is defined as the percentage change in the quantity of water demanded that flows from a one percent change in the price of water. Thus, if the own-price elasticity of demand for water is minus 0.1, this means that if the price of water were to increase by one percent, the quantity demanded would decrease by 0.1 percent. If Q_W is used to denote the volume of water consumed, and P_W is used to denote the price of utility provided mains water, then the mains water own-price elasticity formula at a specific point can be given as:

$$\eta_{Q_w,P_w} = \frac{\partial Q_W / Q_W}{\partial P_W / P_W} = \frac{\partial Q_W}{Q_W} \times \frac{P_W}{\partial P_W} = \frac{\partial log Q_W}{\partial log P_W} = \frac{\text{percentage change in } Q_W}{\text{percentage change in } P_W}. \quad (1)$$

In terms of interpretation, if the own-price elasticity of mains water is less than minus one, the demand for mains water is said to be price elastic. A value of less than minus one would mean that the volume of water demanded is relatively sensitive to price changes. If the own-price elasticity of mains water is greater than minus one, the demand for water is said to be price inelastic. In practice this means that the volume of water consumed is not sensitive to price changes. By the law of demand there is a non-positive relationship between price and quantity so that the own-price elasticity values must be non-positive: i.e. lie between zero and minus infinity.

The cross-price elasticity of a good measures the percentage change in the quantity of a good — say recycled wastewater — demanded as a result of a one percent change in the price of a different but related good, say the price of mains supplied water. The key difference between an own-price elasticity measure and a cross-price elasticity measure is that for the own-price elasticity measure the price and quantity relate to the same good (e.g. the price of mains water and the quantity of mains water) whereas for the cross-price elasticity the price and quantity relate to different things (e.g. the price of mains water and the quantity of recycled wastewater).

If the cross-price elasticity of demand for mains water and recycled wastewater is 0.05, it implies that if the price of mains water were to increase by one percent, the quantity of recycled waste water demanded would increase by 0.05 percent. Where the cross-price elasticity is positive, the goods are referred to as substitutes. Where the cross-price elasticity is negative, the goods are referred to as complements. The formal result for the cross-price elasticity of demand between mains water and recycled wastewater, where Q_R denotes the quantity of recycled wastewater demanded, and P_w is the price of mains water is given as:

$$\eta_{Q_R,P_W} = \frac{\partial Q_R/Q_R}{\partial P_W/P_W} = \frac{\partial Q_R}{Q_R} \times \frac{P_W}{\partial P_W} = \frac{\partial log Q_R}{\partial log P_W} = \frac{\text{percentage change in } Q_R}{\text{percentage change in } P_W}. \tag{2}$$

The fundamental economic theorem of demand homogeneity requires that the (Hicksian) own-price elasticity of a good, plus all the relevant cross-price elasticities, must sum to zero. Using the above notation this theoretical requirement can be expressed as:

$$\sum_j \eta_{Q_i,P_j} = 0, j = (1, \dots, n). \tag{3}$$

Demand homogeneity therefore tells us that own-price elasticity of mains supplied water (or any other good) is determined by: (i) the number of substitutes, and (ii) the extent to which products are substitutable.

This insight about determinates of the own-price elasticity of demand is important, and has the following implications:

- if the alternatives to using mains water are limited, changes in mains water charges will have little impact on the volume of water consumed
- if alternatives to using mains water are introduced, the price elasticity will change. That means that non-price related policies that increase the number of substitutes to using mains water will impact price responsiveness
- if the price signal is weak, for example due to the use of block pricing where consumers face only an annual water bill, responsiveness to price changes is likely to be low
- if, for some water based activities, say outdoor use of water for maintaining a garden, there are more alternative options to using mains water than in other situations, such as within home use, then the quantity responsive across activities where there are more substitute options will be greater than for cases where the substitute options are very limited

- as the scale of the price increase considered increases, the alternatives to mains supplied water that are economically viable will increase. As such, price responsiveness could be very low for modest price changes, but become higher as each new substitute technology crosses the threshold of economic viability. This means that the price elasticity may not be constant

- it can take time to understand the alternative options available following a price change, and it can take time to build the infrastructure required for substitute options to be implemented. The immediate, or short-run own-price elasticity is therefore likely to be more inelastic (less price responsive) than the long-run own-price elasticity.

The available empirical evidence on consumer behaviour regarding water demand is consistent with these *a prior* expectations (Jenkins et al. 2003; Scheierling et al., 2006; Young, 2005). A final complication with water supply assessment is that consumers often don't understand the extent of their water consumption (Beal et al., 2011). When consumers do not understand their own water use it is difficult for them to understand how they can change their use patterns.

Existing water supply literature

Evaluations of the value of additional water supply have mainly focused on the benefits of avoiding government imposed water use restrictions during periods of water shortage; and improvements in water quality and service reliability.

Averting behaviour studies

Powell (1991, documented in NRC 1997 page 90) studied 15 communities in Massachusetts, New York, and Pennsylvania, and found residents who were aware that their water supply was contaminated by trichloroethylene or diesel fuel, spent on average US$32 per household per year on bottled water. This expenditure was four times higher than the spending on bottled water of those living in uncontaminated areas.

The averting costs associated with avoiding Giardia-contaminated water from a community water system in Pennsylvania, USA were estimated in Laughland et al. (1996). The averting costs were defined to include the opportunity costs of time to boil or haul water, and the direct costs associated with purchasing clean water, and were estimated to be $14.14-$36.33 per month per household.

For the Korean context, Um et al. (2002) estimated citizens' WTP to improve their tap water to different quality levels. The authors extended the conventional averting behaviour method into a perception averting behaviour method for valuing different pollution levels of tap water by investigating different types of drinking water and different perceived pollution level of tap water quality. Depending on household income level, the estimated minimum WTP value was found to be $4.20 -$6.10 per month per household.

Rosado et al. (2006) used both the averting behaviour method and the contingent valuation method to estimate WTP for drinking water quality in urban Brazil. The estimated WTP for treating tap water to a drinkable standard was $5.20 to $19.50 per month, per household, in addition to existing water bills. The authors argue that using a combination of different resources and datasets results in the estimation of robust WTP values. The authors also note that unless careful consideration is given to issues such as heteroscedasticity, estimates will be biased.

A case of groundwater contamination is considered in Abdalla (1990). Specifically, the study considers the averting behaviour costs of residents in a region in Central Pennsylvania, USA, where the local groundwater source was contaminated. The extent of local concern about the issue is reflected in the survey response rate. Out of a total resident household population of 1,596 the authors received 1,045 completed surveys. The study found that the cost of residents' averting behaviours, such as boiling water and buying bottled water were about $252 to $383 per household per year.

Pattanayak et al. (2005) used the averting behaviour method to estimate the averting expenditure by households in Kathmandu, Nepal, where residents only have access to an unreliable flow of poor quality water. The averting behaviour considered included pumping water from springs and deep tube wells, purchasing water, and storing and treating the poor quality water that was supplied. The results showed mean monthly household averting expenditure (including collection costs, pumping, treatment, storage, and purchase costs) was around $3. Averting expenditure was, however, also shown to vary with household income, and the mean value of monthly averting expenditure for poor households was around $1.4.

A common feature of the above research is that it relies on costs (or opportunity costs) that actually occur to estimate the value of water resources. Intuitively this makes the results seem more reliable than results derived from hypothetical scenarios. There are, however, a number of issues that can lead to biases in averting behaviour studies. First, people may continue to purchase bottled water even though the tap water has improved to drinkable quality. This would lead to an over-estimate of the averting behaviour costs. Second, as averting behaviour focuses on costs rather than benefits, the values may only represent a fraction of the benefits. Third, alternative water resources may not be available. For example, it may not be convenient/ possible to buy bottled water even though the residents want to do so. A final limitation is that the method is really only useful for considering changes such as raising water quality from below drinking standard to drinkable standard.

Contingent valuation studies

In many developing countries the majority of houses do not have private connections to mains water and only public taps are available where access is shared by households. To use water from public taps there are opportunity costs in terms of the travel time required to collect water. In such scenarios contingent valuation studies can provide useful information regarding the amount communities would be willing to pay to have improved water supply services, such as an individual house connection. For households to be able to use water from private connections there are generally both charges for the connection, and for the water used.

Whittington et al. (1990) is a contingent valuation study undertaken in Southern Haiti. Based on a total of 170 completed questionnaires the study found that people would pay 1.7 percent of their monthly household income to have a public standpost near their homes, and would pay 2.1 percent of their monthly household income for private connections in their yards.

As it considers responses from the same people before and after an actual intervention, Griffin et al. (1995) is an interesting contribution to the contingent valuation literature. The surveys were conducted in the Indian State of Kerala in an area where there were saline issues with the local water supply. The first survey was conducted in 1988 and was to estimate residents WTP for improved water services. The second survey was conducted in 1991 after a new water supply system became available and aimed to investigate whether residents' actual behaviour was consistent with how they said they would behave in relation to connecting to the water supply system. Although specific details were not reported, the general finding was that residents' stated behaviour did not match their actual behaviour.

In developing countries, household income, access to water connections, and the quality of water services etc. can influence people's WTP for water supply services. This in turn can make it difficult to establish a single representative WTP value from any given study. Briscoe et al. (1990) estimated the willingness to pay for water supply services in three areas in Brazil focusing on estimating the income and price elasticity of demand. Results show that the average stated maximum willingness to pay to have a connection to private yard taps was around 100 cruzados per month. At the time of the study this amount was 2.5 times higher than the actual monthly tariff.

Altaf et al. (1993) investigated the WTP of households in the Punjab region of Pakistan. The study found that households without piped water connections would like to pay Rs.56 per month (4.7 times higher than the monthly tariff at the time) for connection to a water system with standard reliability. Those who already have piped water systems would be willing to pay an additional Rs.33 per month (2.8 times higher than the monthly tariff at the time) to have adequate water supply pressure.

The WTP of households for improved water services in Kathmandu, Nepal was investigated in Whittington et al. (2002). The study relied on 1,500 survey responses. The question of interest was how much households would be willing to pay for services from a private service operator. The private operator could provide services such as improved water quality and decreased frequency of water supply interruptions. For households already connected to water supplies provided by public operators, which only provide water for a few hours per day with low pressure, the average monthly WTP per household to be connected to water provided by private operator was US$14.3. This value was equal to 6.3 percent of average household monthly income. For households that currently have no water connections, their mean monthly WTP per household was US$11.67, and for these households this represented 5.1 percent of average monthly income.

Devoto et al. (2012) found that households in urban Morocco would be willing to pay almost double their current water bills on private water connections at home, versus $US11 per month for a public connection close to their homes with the same level of water quality. The existing costs are the fees paid to their neighbours who have water connections to access water and the time costs to collect water from public connections (they spent nearly 18 hours per month for collecting water from public connections on average). Without improved water quality and quantity, the benefits from new installed private or public water connections seem to be a function of the time saved.

In developed countries, as most houses are connected to a water supply network research has focused on water quality, water service reliability and water resource protection issues. For example, Carson and Mitchell (1993) estimated the national benefits of freshwater protection in the USA. Water quality was defined in increasing levels of quality as: fit for boating activities; fit for boating and fishing activities; and fit for boating, fishing, and swimming activities. Based on 813 survey responses the study found that the annual mean WTP per household to keep freshwater resources at a quality level suitable for: boating activity was $93; boating and fishing activity was $163; and boating, fishing and swimming activity was $241.

The WTP of Canadians to support a program to repair water distribution and sewage treatment systems to prevent a decline in current water services was investigated in Rollins et al. (1997). Based on 1,511 household surveys across Canada the study estimated that the mean WTP to support a program to repair water distribution and sewage treatment systems to prevent a decline in current water services was about CA$26 per month in addition to household current water bills. The study claimed that as the differences of WTP among Canadian regions were not significant, the results of the survey can be used to estimate the WTP of the whole nation. On this basis the national WTP was estimated as CA$1.1 billion less than the amount required to cover the estimated marginal costs of maintaining, renovating, and upgrading water infrastructure to ensure adequate water services.

The WTP of residents in ten districts in California, USA to avoid water shortages was investigated in Koss and Khawaja (2001). Through the use of 3,769 completed survey the authors were able to establish that residents were willing to pay US$11.61 per month per household to avoid a 10 percent shortage once every ten years; and US$16.92 per month to avoid a 50 percent shortage occurring every twenty years.

Epp and Delavan (2001) investigate household WTP for a proposed groundwater nitrate pollution reduction programme in Pennsylvania, USA, and found that the WTP ranged from US$51 to US$74 per year, depending on whether an open-ended format or a dichotomous choice format was used when surveying households. More generally, the authors note that residents’ WTP for water quality or reliability of water supply services are influenced by many factors in addition to the question format used, including: household income, perceived effectiveness of the programme, expenditure to avert pollution, number of children in the household, gender, and age.

Poe and Bishop (2001) is a contingent valuation study concerning protecting groundwater supplies from nitrate contamination in Wisconsin, USA. The study found that the behaviour of respondents, and their willingness to pay, was influenced by awareness of the safety risks associated with the current water supply. Those who were aware of the risks and used adverting measures such as purchasing bottled water for drinking were generally willing to pay more for water quality improvements. However, the research also found that the WTP for improvements in water quality of those in areas where contamination levels were very high may be lower than the WTP of those unaware of contamination issues. The authors' explanation of this result is that residents in areas of heavy contamination may consider a small reduction of pollution as incapable of bringing a heavily polluted water resource back to safe conditions.

Genius and Tsagarakis (2006) investigated residents WTP for improvements in water quality in the Heraklion area of Greece, an area where water supply disruptions happened regularly, and where many households had refused to drink tap water because the tap water was believed to be contaminated. The authors found those who had problems with the smell or colour of the tap water, or those who had stayed in the city for a long time, were relatively less likely to drink tap water directly. Based on 294 survey responses the estimated WTP of residents for a proposed plan to improve water services such that flows were regular and the quality of tap water was drinkable was €13.8 per month in addition to their monthly bill. In subsequent work Genius et al. (2008) concluded that female respondents, households with higher incomes, households with children, and residents who normally did not use tap water for drinking, were, on average, willing to pay more. This work was based on residents in the Greek town of Rethymno, and relied on 306 completed household level survey responses.

Hurlimann (2009) conducted a survey on WTP per kilolitre (kL) of water among office workers in Bendigo bank head office, Australia in February 2007. This study draws our attention for the following reasons:

- The survey was conducted during a period of extreme water shortages in Victoria. Melbourne dam water storage was around 25 percent, and in Bendigo the situation was much worse. In 2007 with the Bendigo reservoir recorded its lowest ever storage level, which was 4 percent, and there were significant restrictions on local government water use to maintain public open green space due to water shortages;

- Because of the water shortage, water was being carted to and sold in the Bendigo region.

The study found a mean WTP of A$7.7/kL based on 305 responses. This value was around six times higher than the price of mains supplied water. The result was, however, within the retail price range for trucked water, which at the time was between A$6.3 and A$17.1/kL depending on water quality and the transportation distance. The research indicated that residents would be willing to pay prices several times higher than normal water price to avoid strict usage restrictions during drought periods. The study also demonstrates that the estimated WTP from studies can be a reasonable representation of the marginal price of water supplies.

The contingent valuation method can also be used to estimate the value of alternative water supplies. The city of Oulu, in Finland, uses groundwater as a drinking water resources, and Tervonen et al. (1994) investigated the WTP of residents for relying on treated groundwater or purifying water extracted from the Oulu River. The authors found that residents were willing to pay €54 per year per household for purified groundwater, but only €51 per household per year for purified river water. However, whether there is a statistically significant difference of residents' preferences for drinking water supply resources was not clear from this research.

Laughland et al. (1996) surveyed 226 households in Milesburg, Pennsylvania, USA. At the time of the survey the local water supply was contaminated with Giardia. The authors found that households were willing to pay $18 per month in addition to their current water bills to connect to an alternative water source that would provide drinking quality water.

The tap water in Mexico is often polluted and unsafe for drinking. With this as the background context, Vásquez et al. (2009) found that residents in Mexico would be willing to pay 92.74 Mexican pesos, which is as much as 77 percent more than their existing water bills for the provision of safe drinking water to their houses.

Choice experiments examples

Blamey et al. (1999) used a multinomial logit model to investigate preferences across 294 households in Canberra, Australia. Residents were faced with choices between using recycled water for outside use, construction of new dams, and water restrictions. Use of recycled water for outdoor use was the highest ranked water supply option among the choices. The mean WTP for the provision of recycled water for outdoor use was A$47 per year. There was, however, also a clear difference in preferences between using recycled water for drinking and using recycled water for outdoor use: residents had a clear preference for avoiding drinking recycled water.

The choice experiments method was used in Hensher et al. (2005) to examine Canberra residents' attitudes towards drinking water and wastewater. Based on 211 completed surveys, the authors found that the WTP of households depended on the way the questions about reliability of drinkable water and wastewater services were set out. Annual mean WTP to reduce the frequency of water supply interruptions from twice a year to once a year was A$41.51 per household. However, if residents currently face monthly interruptions, the mean WTP to reduce the water supply interruptions to bimonthly is only A$9.58. Households' WTP to reduce wastewater flow from twice a year to once a year was estimated to be A$77.85, and for reduced wastewater flow from once per year to once every two years was estimated to be A$116.77.

Choice experiments were used in Tapsuwan et al. (2007) to assess the preferences of residents in Perth, Australia for water resource development options to avoid outdoor water restrictions. At the time of the survey residents were faced with restrictions on the outdoor use of water. Based on 414 completed surveys, the results showed that residents would be willing to pay 22 percent more on their annual water usage bills to be able to use their lawn and garden sprinklers on three days per week rather than one day per week.

Hedonic price studies

The extent of hedonic price studies considering water supply issues is limited. Connections to a mains water supply network are, however, still an issue in some developing countries and whether a water connection is available or not can affect the rental price of a house. Several studies have looked at this issue. In the context of Manila in the Philippines, North and Griffin (1993) examined the rental price difference for homes with and without a water connection and found that housing rent would increase by about 30 pesos per month, on average, when a water connection was available. Komives (2003) considered the issue in Panama City and found that an in-house pipe connection resulted in an increase of about $US22 per month in house rent. Finally, Alam and Pattanayak (2009) found that household, in the slums in Dhaka, Bangladesh, with piped water had rental prices that were about US$10 per month higher than houses without a piped water connection.

Where water connections are not always a standard feature of homes, having a water connection can also affect the property price. Nauges et al. (2009) studied the property market in Central American cities using the hedonic price method and found that a tap water connection added between 10 percent and 52 percent to house prices.

Summary: water supply issues

As can be seen from the studies reviewed, a number of different approaches have been used to investigate the value of water supply to households, and these methods are all reasonable. The limitations to the existing work do, however, need to be noted. Implied values tend to vary with approach, which is a concern. Further, within a given approach, it is also the case that there are differences in values depending on factors such as household income, gender, number of children in households, and culture. Values can also vary significantly depending on people's awareness and understanding of current water supply service quality. It is difficult to capture all these differences in a single study and this in turn means that reported results may not capture the complete picture.

An important aspect to consider when discussing the existing literature is the transferability of the results. The estimated values may be localised and it may only reflect the value of a particular service at a particular point in time. According to Brouwer (2000), the transfer errors from unadjusted unit value transfer can be as high as 50 percent, and the transfer error can be more than 200 percent in the case of adjusted value transfers. It is therefore important to spend considerable time working through whether or not it is appropriate to transfer specific results to new locations.

A recent trend in the literature with respect to transferring values from one specific study to another location is to combine the benefit transfer method with meta-analysis information (Rosenberger and Loomis, 2000; Shrestha and Loomis, 2001). Meta-regression analysis in particular can be used to synthesise existing research findings when there are many varying study attributes (Glass et al., 1981). The technique can be used to develop a benefit transfer function that takes into consideration more than one study, and is able to provide more robust estimates of transfer values that in turn reflect a more detailed understanding of the differences among individual sites and resources (Shrestha et al., 2007). Validation tests of this combined approach are, however, still required to ensure method validity.

Pollution and flood hazard reduction

In urban areas, stormwater runoff can cause sudden increased pollutant levels in surface waters which can lead to significant negative impacts on ecosystems and the environment (Roy et al., 2008). The large amount of stormwater within a short time period can also put great pressure on urban drainage systems, which may increase the possibility of the occurrence of flooding.

Nonpoint source pollution of stormwater

Since at least the 1970's it has been understood that urban stormwater runoff contains pollution components (Barton, 1978). These pollutants are believed to be washed off from car parks, lawns, roads, and highways; and this type of pollution is referred to as nonpoint source pollution (Bourcier et al., 1980, Hoffman et al., 1985). With the worldwide awareness of the need to protect the environment, major point source pollution is gradually being eliminated, and in some cases nonpoint source pollution is now the dominant pollution type in urban water systems (Petrone, 2010). The main contaminants in urban water runoff include: sediment, nutrients, pathogens, and chemicals (Makepeace et al., 1995). These contaminants enter water bodies from flows carried along the stormwater drain network, or seep into the groundwater and transfer into main streams with groundwater movement.

Initial economic valuation studies on nonpoint source pollution largely focused on estimating the damage costs caused by the pollution and/ or the environmental and public health risks created by pollution (Philips, 1988; Haynes and Georgianna, 1989). As it is hard to separate the influence of point source pollution from nonpoint source pollution, initial economic evaluation studies tended to estimate the impact of different pollution sources as a whole. For example, working through an extensive economic analysis process, Farber (1992) estimated that the costs of the environmental risk caused by both point and nonpoint source pollution in the USA could be as high as 2.7 percent of GDP.

In terms of understanding the nonpoint source pollution problem, Ventura and Kim (1993) suggest that urban nonpoint source pollution can be understood as a function of land uses (such as the amount of impervious surface), land use associated contaminant sources (such as vehicles, industrial debris, leaf and animal litter, etc.) and other physical properties of the land (such as slope, soil structure, and hydrological and meteorological characteristics of an area). Therefore, for urban areas, the empirical models used to estimate pollutant load are primarily driven by land use related data.

Geographic Information System (GIS) is an ideal tool to process land use data accurately. Tsihrintzis et al. (1996) provide a literature review that summarises the fundamentals of GIS and discussed the application of GIS to water resources management, including nonpoint source pollution and flood prediction.

GIS also makes it possible to develop hydraulic models that simulate the actual pollution transfer speed and pathways. For example, Lai et al. (2011) developed an integrated two-model system which contained one multimedia watershed model and one river water quality model to simulate the impacts of nonpoint source pollution on the water quality of rivers in Taiwan. The land use patterns were classified using SPOT satellite images and Digital Elevation Model techniques with the aid of the ArcView GIS system. The nonpoint source pollution loadings of the Kaoping River Basin (Taiwan) were calculated using this integrated system, and changes in land use patterns can then be directly linked to water quality changes with consideration given to nonpoint source pollution load.

Combining economic valuation methods with GIS techniques for an urban water setting is something that does not yet appear to be a feature of the literature, although there are examples of this kind of integrated research for rural areas (Franco et al., 2001; Merem et al., 2011). There are several possible reasons for the lack of focus on the urban setting. First, it could be that in urban areas the nonpoint source pollution types, demographic compositions, geology and hydrological conditions are all much more complex than in rural areas. Second, it could be the case that the nonpoint source pollution problem is more pronounced in agriculture dominant rural communities.

The knowledge base regarding the application of economic instruments to control nonpoint source pollution is well developed; with key approaches including: tax and subsidy strategies, standards and liability rules, contracts and bonds, and emission trading methods (Shortle and Horan, 2001; Xepapadeas, 2011). Existing research, however, focuses on how these instruments can be applied to agriculture nonpoint source pollution problems. How economic instruments can be used in conjunction with stormwater management approaches to reduce nonpoint source pollution in urban areas is not yet a feature of the literature.

Flood hazard reduction

From the catchment level perspective there are two types of flood hazard: urban flooding and rural flooding. Both stormwater and mainstream flow can contribute to each type of flood. The relationship between urbanisation and stormwater flood risk is quite direct. Urbanisation involves paving parts of the watershed with asphalt, straightening and shortening water flow paths by conveying runoff through drainage systems, and the erosion of downstream channels (Parker, 2000). The stormwater collection system can then be overwhelmed, and consequently the areas serviced by the system may be subject to flooding. In terms of the relative importance of stormwater and mainstream flow to flooding, SCARM (2000) report that urban flooding caused by stormwater overflow, on average, represents 11 percent of flooding costs in Australia.

Evaluate flood damage

Estimation of ex post costs can be a direct way of evaluating flood damages, and historically government authorities have counted and recorded flood damage losses after each flood event. These historical data can be used to generate estimates of the potential flood damage risks in certain areas (Thompson et al., 1997). Lovelace and Strauser (1994) reported the flood damage costs of flood events in the Mississippi river basin in 1993 by using expenditures on cleaning up and repairing the levee damages caused by flood. FEMA (2012) estimated costs caused by flooding by adding up the direct losses of individuals, companies, and communities from the event. However, these financial losses cannot be considered as economic losses. For example, one company which is closed for several days because of a flood event may suffer lost profits, but other companies may gain extra profit due to additional sales that previously went to the closed firm. Similarly, losses from disruptions to the road network may, in the end, deliver greater profits to airline and marine transport companies.

Another method that can be used to estimate costs relies on the use of Stage-Damage Curves. This approach, according to Smith (1994), can be implemented as follows:

- Select the individual land use categories for analysis;
- Identify the main characteristics of a flood (such as depth, duration, velocity, and load);
- Within each land use category, identify significant subgroups of building types (such as one or two storey houses, houses with a basement etc.);
- Use the main characteristics (or variables) of the flood to establish relationships between the variables and damages (such as deriving a depth damage curve) for each land use subgroup;
- Use the other flood characteristics, such as velocity, to modify the base curve. For example, the stage-damage curve could have low, medium, or high velocity variants.

With the assistance of GIS methods and hydrologic modelling techniques, it is then possible to build flood damage assessment models to evaluate the damages caused by flood events. Existing models of this type include the HAZUS model from the USA (FEMA, 2012) and the NHRC model (Leigh and Kuhnel, 2001) developed by Macquarie University in Australia. Both of these models are capable of generating stage-damage curves which can be used to estimate the damage costs by floods under various conditions.

Evaluate flood risks and protection measures

There is some literature that tries to estimate the value of flood risks through multiplying the estimated flood damage costs with the reduced possibility of flood risks. For example, Blong (2003) multiplied construction costs per square metre with different level of flood risks to calculate the damages to buildings from flooding in Australia. Seifert et al. (2010) used industrial and commercial asset values to estimate losses from potential flood risks in an industry zone in Germany. Estimated values from this type of approach are more closely related to the costs of flood damages rather than benefits of the flood control measures.

Hedonic price studies

The hedonic price method has been used to measure the benefits of flood risk control measures. Properties may sell for a lower price if buyers are aware of the flooding risks of that property.

Although no specific monetary values were reported, Bartosova et al. (2000) found increases in food risks could decrease the value of residential properties within the 100-year floodplain in Wisconsin, USA.

The property value changes in the USA following urban stream restoration measures, including flood protection measures, are calculated in Streiner and Loomis (1995). The authors found that flood damage reductions and stream stabilizations together can add around 3 percent to 5 percent to the value of properties. Note, however, that from the information contained in the paper it is not clear exactly how specific values were obtained.

The hedonic price method is used in Harrison et al. (2001) to estimate the housing discount for homes in the 100-year flood plain. The data for the study relate to the period 1980-97 and are for Alachua County in Florida, USA. The discount for being in the 100-year flood plain was found to be around $3,000. The authors also note that the net present value of the additional insurance premiums associated with a home on the 100-year flood plain are more than the discount in the capital price of a home on the flood plain.

Insurance costs

In terms of using insurance costs as a measure of flood costs, Chivers (2001) argues that insurance expenses may fail to accurately predict potential flood damage risks as people under-estimate flood damages before a significant flood event, and overestimate risks after a flood event. For example, Bin and Polasky (2004) compared house price differences pre- and post-hurricane Floyd for homes on the flood plain in Carolina, USA. They found that the house price discount doubled within flood zones after hurricane Floyd. This discounted price was also significantly higher than the net present value of the additional insurance premiums. This means residents would be willing to pay a much higher value to avoid flood risks than the actual required insurance fees.

Contingent valuation studies

There are a number of potential issues with the use of the contingent valuation method to evaluate flood control measures. First, people may not really understand what kind of flood risk they are facing and how the proposed control measures could help them. Second, some residents may have difficulties in understanding technical flood terminology. For example, people that have experienced a flood twice in five years may find it difficult to reconcile their experience with a statement that they are on a one in 50-year flood plain. Thus, a reduction of flood risk from once per 50 years to once per 100 years may not make much sense to some people asked to complete a survey. Third, flood control measures such as dams are multifunctional, and it is hard to disentangle the support that is directly related to the flood control element from the overall support for the project. Despite these potential issues, there have been a small number of attempts to evaluate willingness to pay for flood protection using the contingent valuation method.

Thunberg and Shabman (1991) use the contingent valuation approach to analyse the determinants of willingness to pay for flood control projects of the residents of the City of Roanoke in Virginia, USA. The analysis was based on a relatively small sample size (74 usable responses), and focused on owners of flood-prone land. The results show that property protection aspects will influence residents' willingness to pay for flood control investment, as well as nonproperty considerations such as reduced psychological stress and reduced community disruptions.

The contingent valuation method is used in Bateman et al. (1995) to estimate the WTP in Broadland, UK for a multifunction project that included a flood control function. Based on 344 responses the mean WTP was estimated to be £21.75 per year per household to build flood defence works.

Zhai and Ikeda (2006) investigated the WTP of residents in Toki and Nagoya cities, Japan to avoid the inconveniences caused by flooding such as evacuations. Based on 1,259 responses the study found that the mean WTP was 1,030 yen/person/night. The authors stated that household income, individual preparedness, and flood experiences played a significant role in determining the WTP value.

Brouwer and Bateman (2005) examined residents' WTP in East Anglia, UK to conserve a wetland that had a flood control function. The study relied on 1,747 completed surveys and found a mean WTP of around £216 per year per resident. In the study the percentage contribution to total value attributed to the flood control function was not separated from the other functions of the wetland.

Estimate the value of flood reduction caused by stormwater harvesting

Conventional stormwater management focused on removing stormwater from a site as quickly as possible to reduce on-site flooding risks (Minnesota Stormwater Steering Committee, 2008). Stormwater harvesting techniques may, however, require stormwater to stay on-site for a certain period of time and then make its way into the groundwater system by some means. This process may increase the flood risk. On the other hand, stormwater harvesting techniques also involve the use of more permeable surfaces which may help reduce both the peak and total volume of stormwater. The overall impact of stormwater harvesting techniques on flood risk is therefore ambiguous.

Some design standards require flood control and stormwater harvesting to be considered separately, for example NHDES (2012) and Sunshine Coast Regional Council (2009). Yet, scientists and engineers have developed integrated systems to ensure that additional stormwater runs into the drainage system if the downward seepage rate allowed for in the stormwater harvesting design is insufficient. Household water tanks may also be a reasonably reliable technology for flood reductions (Tam et al., 2010). Overall, however, the effects of collecting stormwater to mitigate flood risks are not clear, and this remains an area where further work is required.

Recharge and improved groundwater quality

Groundwater refers to water stored in underground aquifers. Groundwater aquifers generally provide high quality water that requires little treatment before use. Groundwater is, therefore, an important source of fresh water. As aquifers are out of sight, groundwater protection is a management area that has not always been a priority. The two main issues in groundwater protection are overdraft and pollution.

Main issues of groundwater protection

From a sustainable management point of view, extractions from aquifer should be less than or equal to the volume that is recharged (Tularam and Krishna, 2009). Groundwater is recharged naturally by rain or snow melt, and by surface water such as rivers and lakes. Depending on specific geology of an area the recharge process may take a long time. Natural recharge can be impeded by human activities such as general development activity, paving, and logging. These activities can result in the loss of topsoil, which can lead to reduced water infiltration, enhanced surface runoff, and a reduced recharge rate. Human activity, such as use of groundwater for irrigation may also lower the water table.

According to Zektser et al. (2005) and Cullen (2006), groundwater overdraft related problems which can result in economic losses include:

- Lower groundwater table;
- Reduced stream flow;
- Land subsidence;
- Saltwater intrusion;
- Drainage of acid sulphate soils;
- Sea level rise.

Each of these issues is further developed below.

Groundwater extraction may lead to a reduction in the regional groundwater table depth (Marshall et al. 2006). A lower water table means more energy is required to extract water from the ground (Wyrwoll, 2012). Further, if the groundwater is lower than the existing water table or beyond the reach of existing wells, the existing wells may need to be drilled deeper in order to extract water.

Groundwater sustains rivers, wetlands, and lakes. Groundwater overdraft reduces stream flow and affects ecosystems related to rivers, wetlands, and lakes. In extreme cases, as Candela et al. (2009) point out, streams can effectively be drained. Vegetation may also have difficulty extracting enough water to survive when groundwater levels are lowered (Zektser et al., 2005), which may cause severe environmental problems. For example, Chen (2003) found that the lower groundwater table in a region on the border of Queensland and New South Wales resulted in the death of terrestrial vegetation cover of the area.

The “effective stress” level in soil is equal to overburden total stress minus water pressure (Verruijt, 2012). If the effective stress level increases, soil becomes compressed. When the groundwater level drops the effective stress level of the soil increases. This in turn causes the ground to settle. Virgin compression, which means the compression level associated with reaching an effective stress level for the first time, involves much higher compression compared with recompression. Thus, once the groundwater drops below its previous historical low level, the ground settlement that takes place may not be recoverable when the groundwater is recharged. This kind of land surface subsidence has been known to cause damage to civil infrastructure and buildings (Nevill, 2009).

In coastal areas, a lower water table may induce seawater reversing flow towards land. Increased salinization of groundwater can also play a negative role in agricultural and cause other environmental problems (Tularam and Krishna, 2009).

Acid sulphate soils are non-toxic when they are below water table. However, when exposed to air due to drainage or disturbance these soils produce sulfuric acid, often releasing toxic quantities of iron, aluminium, and heavy metals. Drained acid sulphate soils are often found in low-lying coastal plains and can result in acidification and pollution of freshwater and estuarine streams (Nevill, 2009; Sommer et al., 2001).

Wada et al. (2010) warn that groundwater overdraft worldwide may cause sea level increases. This is because the water that was previously trapped underground is now going into the ocean. Meanwhile, by increasing the amount of moisture available to fall as precipitation, severe weather events are more likely to occur. To some extent, the moisture in the atmosphere may accelerate the probability of a global warming event. However, the correlation coefficient between groundwater depletion and sea level rise is not yet scientifically determined.

Pollution is the other main concern for groundwater protection. In urban areas, pollution sources include nonpoint source pollution caused by urban land users, and point-source pollution, which comes from industry, and other contaminated sites. As the urban area and rural area may share a common aquifer, contamination caused in non-urban areas such as agriculture and mining may also contribute to pollution issues in urban water supplies. The key locations where pollutants tend to enter groundwater systems include coastal areas, catchments, and recharge areas and wetlands on shallow aquifers.

In coastal areas aquifers are frequently threatened by seawater intrusions. Aguilera-Klink et al. (2000) list the major problems associated with seawater intrusions and concluded that improving water demand management and reforming water rights can be effective approaches to management, although the study did not report on the costs and benefits associated with different options. Zekri (2008) examined the economic implications of seawater intrusions. By comparing the different options for reducing extractions, the author concluded that demand management through water pricing and changing water supply to alternative sources could be possible management approaches.

Economic valuation of groundwater protection

Direct use values of groundwater

Economic valuations of the use value of groundwater focus on the role of groundwater as a water supply source (see Table 2 for available literature). There are, however, other use values of groundwater, such as groundwater thermal/cooling systems which use deep groundwater to bring heat into the house and shallow groundwater to put the heat back into the ground. As these other values are minor compared to the role of groundwater as a water supply source they are not reviewed here.

Non-use values of groundwater

Non-use values include option value: the value that the groundwater resource is not currently used but may be used sometime in the future. There is also existence value, which is the value associated with preserving the groundwater resource as it currently is with no intention to use it in the future. The two other non-use values identified in the literature are altruistic value — which is the value obtained by person i from use by person j, where $i \neq j$, and the bequest value — which is the value associated with leaving the resource for future generations.

Because these values are quite hard to quantify, and because they are not linked to any tradable goods, only stated preference methods are able to estimate these values. There has been only limited research of the non-use value of groundwater. Sun et al. (1992) used the contingent valuation method to estimate the option price of groundwater quality protection. In the study option value is used to measure the benefits of groundwater contamination abatement, and it is the individual’s maximum WTP to keep the option to use this resource in the future. The study found the mean option price of groundwater protection from contamination to be $641 per year per household. Authors of early research, such as McClelland et al. (1992) took non-use values such as bequest value as total non-use values. Wright and Hudson (2013) assumed the environmental benefits as the total non-use values. However, the environmental benefits not only contain non-use values but also contain some use values. More generally, it may be hard to separate indirect use value and non-use value for groundwater. For example, reserve groundwater may contribute to plant growth and these plants may in turn provide people with a unique recreation place.

Wastewater management

Globally, around 90 percent of wastewater produced remains untreated, with this wastewater directly recharging rivers and oceans, and potentially causing widespread water pollution (IWMI, 2010). Treated wastewater can, however, be reused by households, industries, agriculture, and natural ecosystems (Daigger and Crawford, 2007). In Australia, although wastewater is treated, only around 10 percent of wastewater is recycled for reuse (Dimitriadis, 2005). With proper management approaches this wastewater could be a valuable and reliable water supply resource.

Wastewater type

Domestic wastewater is usually classified as either grey water or black water. Leftover water from use in the shower, washing machine, hand basin, etc. is grey water. Black water is usually defined as toilet wastewater. As the water has higher load of chemicals, fats, and other organic matter, water from dishwashers and kitchen sinks is often referred to as dark grey water and is normally excluded from grey water reuse (Birks and Hills, 2007). In traditional practice, both grey and black wastewater are combined and removed from a residence using a shared sewerage system. Sewage water is then treated to limited pollution and health risks, before finally being returned to the environment.

Sewage water needs to be treated before reuse. Although in some developing countries sewage water has been used directly without treatment as an irrigation water resource for centuries (Ayres et al., 1992). This practice has, however, been found to have significant negative environmental effects (Qadir et al., 2010). Grey water contains less pollutants and is much easier to treat for reuse than the sewage water. Thus, it may be appropriate to collect, treat, and reuse grey water locally rather than send it to the sewage system to be treated in the main centralized wastewater treatment plants.

The cost of collecting grey water separately is relatively low. With minimal plumbing augmentation, or even just the use of a bucket, grey water can be collected. The collected grey water can then be used directly for gardens or flushing toilets without any additional treatment (Ghaitidak and Yadav, 2013). However, in practice the direct use of untreated grey water has some risks. First, there are sanitation issues and health risks if the water is kept for more than 24 hours so the water must be used within a relatively short time period (Eriksson et al., 2002). Second, the use of untreated grey water in the garden may be harmful to some vegetation, and may not be especially environmental friendly.

Wastewater treatment systems

Wastewater treatments for single allotments are referred to as on-site wastewater treatment (decentralised wastewater treatment) systems. There are many types of on-site wastewater treatment devices and systems available in the market. These systems can be simple or quite complex, and for complex systems there is a risk of misuse or incorrect use. To minimise these risks government agencies provide guidelines and regulations for the household use devices and systems. For example, Environment Protection Authority Victoria provides guidelines for on-site single residence wastewater treatment systems with a capacity of up to 5,000 L/day (EPA, 2013). The guidelines categorise available devices and systems in the market and classify them based on their functions and after-treatment water quality.

On-site wastewater treatment devices and systems are relatively expensive. In part the high unit cost reflects relatively high fixed costs for systems and small volumes; but the more important reason is that residents normally lack the time, knowledge, and skills to maintain systems themselves (WSAA, 2009). Compared to household based systems, the unit cost of treating wastewater in groups of dwellings, or at the community scale, could be much less. Community level systems are generally called cluster decentralised systems.

Regardless of whether the system is a single residence decentralised system or a cluster decentralised system, it is recommended that the reused water be limited to light grey water to keep treatment costs down and limit potential risks to the individuals (Friedler and Hadari, 2006). Centralised wastewater treatment systems, on the other hand, do not generally separate grey water from the sewage and treat the sewage water as a whole. Centralised systems may consist of conventional or alternative wastewater collection systems, and centralised water treatment plants and systems to dispose or reuse the treated effluent (Tchobanoglous et al., 1998). In urban areas it is likely to be more cost effective to integrate decentralised systems into the overall wastewater treatment network (Molinos-Senante et al., 2010).

Various techniques have been developed by scientists and engineers to treat wastewater. These techniques includes: use of constructed wetlands (Halalsheh et al., 2008); filtration methods (Gross et al., 2007; Finley et al., 2009); rotating biological contactor methods (Pathan et al., 2011); use of a membrane bioreactor (Arceivala and Asolekar, 2007), and use of an up-flow anaerobic sludge blanket reactor (Hernandez-Leal et al., 2010). The efficiency of these systems depends on factors such as humidity, temperature, source of water, and the requirements of the end use water products. For example, Hellstrom and Jonsson (2003) compared 14 different treatment plant for wastewater from single houses at a location 35 km south-west of Stockholm. The study concluded that different systems perform differently in terms of their efficiency in removing particular kinds of pollutants. Scale also matters, for example Humeau et al. (2011) compared two different systems for 50 and 500 households and found the efficiency of the two systems varied across the treatment scales.

Economic valuation of wastewater treatment systems

There are papers that present comparisons of the benefits and shortcomings of decentralised and centralised wastewater systems, for example, Ho and Anda (2004); Livingston et al. (2004); and Li et al. (2009). These comparisons, however, do not emphasise economic aspects, which are the focus here. For decentralised wastewater treatment systems, the direct costs identified in Lechte et al. (1995) include:

- Grey water collection costs: network separation may need to divide grey water from black water, and an additional pipe line may be needed to supply the treated grey water for reuse;
- Device and installation costs;
- Operation and maintenance costs.

Possible indirect costs identified in Friedler and Hadari (2006) that need to be considered with decentralised systems include:

- Increased probability of blockages in sewage systems due to lower flows;
- Higher pollutant load in the sewage system as the less polluted grey water is recycled and reused.

Risks associated with decentralised systems need to be considered as well as costs, and risks identified in Eriksson et al. (2002) include:

- Financial risks: the system built may not be suited for the application because the situation has changed. As the investment required for a decentralised system is much less than a centralised wastewater treatment plant, the financial risk for a decentralised wastewater treatment system is much lower than for a centralised system;
- Health risks: depending on the source of water, and the treatment methods used, the water available for reuse may still contain chemical and microorganisms that may be harmful to residents' health. Accidental misuse, especially by children, is also a concern (ACT Government, 2007);

- Environmental risks: grey water users need to be careful in the selection of detergents to minimise environmental risks (EPA, 2013). For example, Howard et al. (2005) found that the household chemicals in grey water could be very harmful to plants in gardens;

- Social risks: there are social and cultural risks associated with using recycled water. Research has shown there is social resistance to grey water reuse, especially when the water is used within the house (Dolnicar et al., 2011). There could also be issues associated with cultural beliefs. For example, in some religions people may believe water bodies have spirits (Morgan, 2006).

The direct benefits of decentralised wastewater treatment systems identified by Friedler and Hadari (2006) include:

- Cost savings on water bills;

- Cost savings on sewage bills;

- The avoidance of restrictions on water use in times of water shortage.

Indirect benefits of decentralised wastewater treatment systems identified in Pinkham et al. (2004) include:

- Increased capacity to provide water supply that closely matches the actual growth in demand for water;

- Postponing the need to develop new water supply projects such as seawater desalinisation plants;

- Reduced total energy consumptions as the distance from treatment to the point of use is shortened;

- Reduced sewage water volumes, which can reduce the need to build large centralised wastewater treatment plant.

There are both social and environmental benefits associated with decentralised systems. The social benefits of decentralised wastewater treatment systems identified in Parkinson and Tayler (2003) include:

- Reduced risk and cost of wastewater system failures;

- Providing an empowering experience for people through the promotion of self-reliance and the principles of sustainable use of water.

Finally, the environmental benefits of the decentralised wastewater treatment system include:

- Reduced volume of water to the sewer and a decrease in the amount of treated effluent discharged to waterways, therefore reducing ecological impacts;

- Increased water supply that can be used to irrigate crops, serve ecosystems, or recharge groundwater.

Case studies of economic valuation

From the above discussion it can be seen that social and environmental values play a significant role in the overall economic valuation of wastewater treatment projects. Thus, non-market valuation methods are usually needed. From the various non-market valuation methods available, the most commonly used method has been the contingent valuation method. Overall, the existing research shows that the public is willing to pay significant amounts of money for wastewater treatment projects (see Table 3 for a summary of the literature that reported WTP values).

Contingent valuation studies

Using the contingent valuation method, Tziakis et al. (2009) estimated residents' WTP for a centralised wastewater treatment plant in northwest Crete. The results showed that the mean WTP for a centralised wastewater treatment plant was €21.02 in addition to their average quarterly drinking water bills.

Gillespie and Bennett (1999) estimated the environmental benefits from two sewage treatment proposals that would reduce the flow of untreated sewage from the Vaucluse area (NSW, Australia) to the ocean. One proposal involved construction of a tunnel and the other construction of a sewage treatment plant. The results showed that the mean, one-off WTP for the tunnel option was $137, and the mean, one-off WTP for the sewage treatment plant option was $76.

Genius et al. (2005) estimated the WTP for a wastewater treatment plant in three locations using the contingent valuation method. The locations were the rural and seaside tourist areas of the Municipalities of Lappaion, Georgioupolis, and Krioneridas in North-West Crete. The results showed that the mean WTP for a wastewater treatment plant was a €44 increase in household quarterly water bills. The study concluded by noting that the WTP value is higher than the investment costs of a wastewater treatment plant.

Kotchen et al. (2009) used the contingent valuation method to estimate the WTP of residents of Santa Barbara and Ventura countries, California, USA for a pharmaceutical disposal program. The program was proposed to solve a problem of pharmaceutical compounds in treated wastewater and in surface water. The results showed that the mean WTP to support the program was $1.53 per pharmaceutical prescription.

Avoiding water restrictions during drought periods is an important factor that contributes to householders' WTP for water services. Dupont (2011) used the contingent valuation method to estimate the WTP of Canadians to use recycled wastewater for toilet flushing as a way to avoid summer lawn water restrictions. The results showed that the mean WTP of households to avoid a 30 percent reduction of summer water use was about $C9.26 per month. Similar research conducted in Bendigo, Victoria, Australia found that households would be willing to pay six times the actual water price for treated grey water during a period of relatively extreme water shortages (Hurlimann, 2009).

Choice experiment studies

The number of studies that have used choice experiments to investigate households' WTP for wastewater reuse projects is limited. Gordon et al. (2001) used this method to estimate the value of recycled water for outdoor use for the residents of the Australian Capital Territory. The results showed that the mean WTP was about an increase in household water costs of about A$47.

Birol and Das (2010) used choice experiments to estimate residents' willingness to pay for improved capacity and technology at a sewage treatment plant in Chandenagore municipality, India. The results show that residents would be willing to pay Rs100.32 per year in addition to municipal taxes for an improved wastewater treatment plant.

Shadow price evaluation method

By using the concept of distance function, the shadow price of environmental goods and services can be calculated. A shadow price is the maximum price that people are willing to pay for an extra unit of a given, limited resources, and this value can also be used in benefit or cost evaluations. More generally, the distance function was developed to evaluate the “difference between the outputs produced in the process under study and the outputs of the more efficient process” (Molinos-Senante et al., 2010).

Hernández-Sancho et al. (2010) estimated the avoided environmental costs from the removal of pollutants from wastewater treatment using the shadow price method. The study includes 43 wastewater treatment plants located in the Spanish region of Valencia. The results showed that the removal of nitrogen and phosphorus through the wastewater treatment process provided the majority of the environmental benefits, and was the function that had the highest shadow prices. This study also found that in terms of nutrient emissions, treating wastewater in wetland areas was far better than discharging wastewater into the sea.

Molinos-Senante et al. (2010; 2011) conducted similar research to Hernández-Sancho et al. (2010) and used the shadow price method to estimate the environmental benefits of improved wastewater treatment based on the distance function of the treatment outputs in the region of Valencia, Spain. The authors concluded that the net profits for wastewater treatment plants were positive, hence the proposed wastewater treatment plants should be considered as economically viable.

Cost-benefit studies

When the costs and benefits have both been estimated, cost-benefit analysis can be used to compare different scenarios. Ko et al. (2004) used cost-benefit analysis to compare the efficiency of using a forested wetland and conventional sand treatment for wastewater. Although both a monetary based approach and an energy based approach are used, the study did not consider the social and environmental costs and benefits.

Godfrey et al. (2009) conducted cost-benefit analysis for grey water reuse systems in residential schools in Madhya Pradesh, India. In this case study, the environmental benefits and social benefits are considered as external benefits. The external benefits were mainly analysed in terms of avoided cost and were mostly based on values from available literature. The results show that the total benefit of grey water reuse is significantly higher than the total cost.

Verlicchi et al. (2012) estimated the costs and benefits for a proposed wastewater reuse project at the Ferrara wastewater treatment plant in the Po Valley, Italy, as a case study. Only financial costs are involved in this study, but the social and environmental benefits are considered and analysed using contingent valuation method. Results show that the proposed projects are financially feasible, as indicated by various economic indicators such as cost-benefit ratio and net present value.

Summary

Although there are still barriers in reusing wastewater, such as the community acceptability issues and the community lacking confidence in wastewater management and regulation (Schäfer and Beder, 2006), generally, residents can accept the idea of using recycled wastewater for non-drinkable purposes, particularly grey water (Gordon et al., 2001).

Centralised and decentralised wastewater treatment systems both play significant roles in wastewater management (Nogueira et al., 2009). Depending on various factors, such as the source of water and the end use quality requirements, the efficiency of different systems and options varies significantly. An economic valuation is therefore required for each individual situation. Engineers and plant operators may only consider the financial feasibility issues when comparing different water reuse strategies or options; hence there is a role for regulators to ensure social and environmental benefits are also considered. This in turn requires regulators and government agencies to have a better understanding of the social and environmental values associated with different wastewater management options.

Ecological and environmental value of water

Categories of values

People rely on ecosystems to provide many water-related services. These services can be organised into five main categories: (i) improvement of water supply; (ii) mitigation of water damage; (iii) enhancement of in-stream water related production, (iv) water associated supporting services and macroclimate effects; and (v) provision of water related cultural and aesthetic services (Brauman et al., 2007).

Water supply improvement refers to increasing the volume and security of the water supply for households, agricultural irrigation, and commercial and industrial water usage. Water damage mitigation includes the reduction of flood damage, saltwater intrusion, groundwater overdraft, water pollution, etc. Economic values from these two aspects have been discussed in detail in the above sections. In-stream water related products and service includes water for hydropower, transportation, recreation and the supply of freshwater products. Water-associated supporting services include using water to support vital estuaries and other habitats, preservation of options for future use, and macroclimate effects (Scheffer, 2005). The cultural and aesthetic services provided by water include the provision of religious, education, and tourism values.

How can WSUD affect these values?

Different beneficiaries from ecosystem services may have different preferences of water. However, there are some common factors relevant to most beneficiaries that relate to water quality, and water quantity and its distributions. These common factors can be influenced by WSUD, as proper water management can improve water quality, increase water quantity, and adjust the water volume distribution in time and space to be more favourable to the production of ecosystem and ecology services.

The effects of water quality are quite obvious; especially to drinking water supplies. However, different beneficiaries require different water quality standards. For example, the requirements of water quality for agricultural irrigation are lower than for drinking water supply, while the water quality requirement for water transportation and hydropower is in turn lower than for agricultural use.

Ecosystems normally need to consume some water (Calder, 1998). Vegetation, plants, animals, and fish all need water to grow. Some activities, such as recreation, may not consume water directly, but still need water bodies to take place, and from these water bodies water naturally evaporates. Other activities, such as transportation, need minimum depths of water (Galil, 2007). The distribution of water volume is also important. Too much water at a given point in time will cause flooding, while too little will cause some vegetation to die. Managing water volume distributions can reduce water damage and have positive benefits.

Economic valuation

From the above discussion it can be seen that there are two ways to evaluate the changes in ecosystem values. One way is to evaluate the changes in each category of value separately; and the other way is to estimate the benefits through analysing the improvement of the main factors, which are water quality, water quantity, and its distributions. The main difficulty with using the first approach is that it may be quite difficult to separate out the different values. For example, people may travel to a lake for fishing in part because they like the scenery of the lake. The main shortcoming of the second approach concerns accuracy. Theoretically, the approach should contain all the ecology values related to these main factors, but people may not realise or understand some values exist, such as natural habitat values from improvements of water quality. In practice researchers have used a mix of these two approaches.

Recreational value

Stated preference methods and the travel cost method are the most commonly used methods for estimation of the recreational value of water.

Kaoru (1995) estimated the recreational benefits of water quality improvements in the Albemarle-Pamlico Estuary, North Carolina USA via 547 survey respondents using the choice experiments method. The study found that the benefits measured varied from $0.09 to $5.16 per person depending on the level of quality improvement.

Parsons et al. (2003) measured the economic benefits to recreation from improved water quality using the choice experiments method in six northeastern states of the USA. In the study separate choice experiment models are used for fishing, boating, swimming, and viewing. The authors found for modest improvements in water quality, almost all the benefits were associated with fishing and swimming. The annual benefit from fishing and swimming were, respectively, about $3 and $5 per person. For significant improvements in water quality, all four recreational activities are associated with benefits, and these benefits were much larger. Swimming and viewing were the activities that showed the highest gains in benefit, respectively, about $70 and $31 per person. For boating and fishing the benefit was about $8 per person per activity. Other studies, such as Parsons and Kealy (1992) and Dupont (2001) have found similar results in terms to the pattern of effects across activities with large improvements in water quality.

Another standard that can be used to measure water quality is clarity. Although water clarity and water quality are not necessarily the same thing, clarity is a term that people may find easier to understand. Marsh and Baskaran (2009) quantified people's WTP for increased water clarity in the Karapiro catchment, New Zealand, using the choice experiments approach. They found that the mean annual WTP per household for water clarity from the current clarity (around 1 meter) to: see up to 1.5, 2.0, and 4.0 meters underwater were, respectively $4.17, $21.03, and $65.82.

Water volume also plays a significant role in recreation activities. Connelly et al. (2007) combined the contingent valuation method and the stage-damage curve approach to explain how the value of recreational boating can be assessed and linked to water levels on Lake Ontario and St. Lawrence, USA. The authors found that as the water level drops, economic losses would be expected because some boats could not get out of their slips. Approximately US$1.7 million in economic benefits would be lost if the water level was 244 feet (74.4 meters) for the entire month of August.

Sale et al. (2009) assessed the amount that recreational users are willing to pay to secure an increase in freshwater inflows into two South African estuaries, the Kowie and the Kromme using the contingent valuation method. The study relies on a sample of 150 respondents at each estuary site obtained during December 2002 to January 2003. The authors concluded that the value of freshwater inflows into the Kowie and the Kromme estuaries were around R0.072/m^3 and R0.013/m^3, respectively.

Some studies have considered changes in water quality and volume simultaneously. For example, Crase and Gillespie (2008) estimated the recreational values of visitors to Lake Hume under different water quality and water level scenarios using the contingent valuation method. The study concluded that the recreational benefits were increased by about $1.3 million per annum when the storage level was increased from 50 percent capacity to near full. The annual consumer surplus derived from recreational users of the lake was reduced by about $1 million in the event of an algal bloom.

Sutherland and Walsh (1985) use the contingent valuation method and show that the recreational value attributed to an asset by households can fall with household distance to the asset. This specific study was based on data from a regional household survey of WTP for water quality at the Flathead River and Lake areas in the USA. Regression analysis was used to estimate the relationship between WTP and distance to the study area. The results showed that the WTP significantly decrease with increase in distance. This phenomenon may be partially due to the travel cost associated with increasing with distance from the asset.

Another way to estimate recreational value is the travel cost method. Fleming and Cook (2008) evaluated the recreational value of Lake McKenzie, New Zealand using the travel cost method. Based on analysis of 1,360 surveys, the authors concluded that the recreational value of the Lake ranged from $13.7 million to $31.8 million per annum or from $104.30 to $242.84 per person per visit.

There are a few studies that combine the contingent valuation method with the travel cost method to estimate recreational values, for example, Huang et al. (1997) and Azevedo et al. (2003). Rolfe and Prayaga (2007) estimated the value of recreational fishing at three major freshwater impoundments in Queensland, Australia, using both the travel cost and the contingent valuation methods. The travel cost method was used to estimate the consumer surplus of recreational anglers, and the contingent valuation method was used to estimate the marginal value of potential improvements in fishing experiences. The authors claim that different non-market valuation techniques are appropriate for different components of the valuation exercise.

Besides these methods, other methods such as dose response method (Soller, 2006) and the medical expenditure and health risk method (Zmirou et al., 2003) can also be used to evaluate the recreational value of water. These approaches are, however, not considered here.

Habitat conservation value

There are economic values in conserving natural habitats. Besides the profit gains from tourism and recreational activities, conservation of endangered animals or rare plant species provides scientific value for current and future research. Commonly seen plant species growing in an unexpected location can also be considered as “rare species” and have high values. For example, mangroves, which are commonly seen in tropical areas like North Queensland, also cover a small percentage of the Victorian coast, and in Victoria mangroves may be considered rare. The uniform low height mangroves at Millers Landing in Corner Inlet, Victoria are known as the world’s highest latitude mangroves. These mangroves also provide coastal protection and scientific value[2].

Possible approaches that can be used to estimate the value of habitats include the contingent valuation method and choice experiments. Nunes and van den Bergh (2001) summarised the methods for evaluating natural habitat and species protection and concluded that monetary valuation of changes in biodiversity can make sense. Farr et al. (2013) summarised studies on non-consumptive use and non-use values of rare or endangered species and found estimated values are particularly sensitive to the questionnaire design. This suggests study findings in this area should be treated with caution.

White (2008) assessed WTP among certified U.S. scuba divers for particular wildlife encounters while diving. The study found that the mean WTP for an increased likelihood of swimming with a sea turtle in the wild was $29.63 per year; for sharks it was $35.36 per year; and for coral it was $55.35 per year.

Ressurreição et al. (2011) estimated the public’s WTP to avoid losses in the number of marine species in the waters around the Azores Archipelago, Portugal. The author found that the mean WTP for visitors to prevent 10 percent and 25 percent loss in numbers was €71 and €83 for birds; €86 and €100 for fish; and €85 and €99 for mammals. In each case the cost was framed as a once only payment.

Johnston et al. (2011) used a choice experiment to investigate the value of species protection in Rhode Island, USA watershed. The research found that a single species increase of freshwater mussels was associated with a WTP of $1.86 per household per year, while an increase in the number of native fish species was associated with a WTP of $1.93 per household per year.

[2] www.mangrovewatch.org.au [accessed 10 December 2013]

Aesthetic value

The aesthetic value and the recreational value of water are different. Although natural beauty is an attraction for people to conduct recreational activities, it is not necessarily the reason people visit a place for recreation purposes. Water has aesthetic value independent of recreation value. Beautiful water bodies are always attractive and can provide people with significant enjoyment. In fact, millions of tourists visit lakes, oceans, streams and waterfalls each year with the main purpose of just experiencing the natural beauty of the water bodies rather than undertaking recreation activities. It is also the case that people are willing to pay high prices for properties near clean and beautiful water bodies and do not want properties near dirty and smelling polluted waterways.

From the available literature, three approaches have generally been used to determine aesthetic values: the Photo-Projective Method (PPM), which asks residents to take pictures of their environment and record their descriptions of each scene on site; the opinion of experts; and the hedonic price method. Note that with the PPM information is obtained on people's preference, but not on monetary values.

Pomeroy et al. (1983) measured the perception of an urban river scape, using unbiased differentiation of riverscape photographs. The study sample was 30 university students in Canada that came from various backgrounds and disciplines. The authors found that the cognitive response to photographic quality was completely overshadowed by the responses to the landscapes in the photographs.

Yamashita (2002) explored adults' and childrens' perception and evaluation of water in landscapes. The author found that if children are the main users of the environment, planners need to focus more on the quality of short-distance elements. Pflüger et al. (2010) assessed aesthetic preferences for river flows in eight reaches on six southeast New Zealand rivers via 449 completed online surveys. The survey results indicated that high flows and minimal bank exposure were preferred in small rivers; and intermediate or low flows and low turbidity were preferred in large rivers.

Water quantity is an important element of the overall aesthetic quality of water bodies. Brown and Daniel (1991) measured people's scenic beauty judgements through the use of video sequences depicting a river at different flow rates. This research found that about 10 to 25 percent of the variance in scenic beauty can be explained by flow rate.

Aesthetic value can also be evaluated via expert or public opinion. Some researchers, such as Tudor and Williams (2003) and Nijnik et al. (2009) have used this approach. However, as earlier work by Hekkert and Wieringen (1996) has pointed out, aesthetic values are different for different people, where it is common for there to be substantial variation between expert and public views.

Using the hedonic price approach, Blomquist (1988) found that people are willing to pay higher price for properties with a water view. Specifically, the study found that households along Lake Shore Drive, Chicago, USA, pay on average, $507 per year to obtain a water view. Further, the influence of water on the property price decreases with distance (Sander and Polasky, 2009). Finally, Fraser and Spencer (1998) found water quality was also a key factor impacting house prices.

Summary

In the existing literature the prominence of different methods for evaluating ecological and environmental values varies depending on the specific aspect under consideration. The travel cost method and the contingent valuation method were both widely used to evaluate recreational values, although the travel cost method was not as popular as the contingent valuation method and the choice experiments method for estimating the value of habitat conservation. The hedonic price method was the most widely used method to estimate the aesthetic value of water.

Generally evaluations were focused on one or two of the key elements that affect the value of water quality, water quantity, and its time and location distributions. To date research has been more focused on water quality issues than water quantity issues. Overall it also remains the case that the ecological and environmental values of water are difficult to evaluate, with the ecosystem benefits provided to human still not well understood. Although the monetary valuation of changes to ecosystem can make sense, as there is uncertainty in the values obtained, combinations of methods, or the use of different methods for cross-checks on results, seems to be necessary. A final issue is that there are interaction effects among values such that estimation of values for individual components may result in the overestimation of values.

Conclusions and recommendations

Adopting WSUD concepts and techniques into existing water management approaches has the potential to provide significant benefits in terms of securing water supply, improving water quality, reducing flood risk, protecting groundwater, and supporting the environment and ecosystems.

Non-market valuation methods are widely used for estimating water-related values. The benefits of different functions have been examined worldwide, and the existing literature provides a reasonable knowledge base for benefit evaluations. The ability to transfer non-market values from one location to another location remains an issue where there is potential for improvement.

Attempts to evaluate the total benefits of a water-related project are rare. Most of the studies that claim to evaluate total benefits have not, in fact, considered benefits in a comprehensive fashion. Some of the studies claiming to consider total benefits ignored social, environmental, and ecological values, and considered the direct use values of water only.

It can be important that policy makers consider social, ecological, and environmental values. Unfortunately, the accuracy and consistency of non-market evaluations is not always robust. The methods adopted can significantly influence the estimated results. Thus, close attention to best-practice guidelines is important.

Non-market values are often public, rather than private values. As a result, beneficial changes in water management and utilisation do not necessarily happen spontaneously. Therefore, proper policies and guidelines are needed to encourage both owners and engineers to make decisions that consider the full range of social and economic costs and benefits.

Table 1: Water supply valuation surveys

Author	Method	Location	No. of completed surveys	Object	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
Whittington et al. (1990)	CV	Laurent, Southern Haiti	170	Estimate the WTP for improved water services (private connection)	6.7-7.5 gourdes per household per month for private connections (accounted for 2.1% of household income)	$2.3-$2.6
Briscoe et al. (1990)	CV	Brazil	400	Estimate the WTP for water services (yard tap)	100 cruzados as the average stated maximum WTP for a yard tap (2.5 times the monthly tariff at the time of survey and accounted for 2.3% of average family income)	$2.8
Howe et al. (1994)	CV	Colorado, USA	588	Estimate the WTP for improved water service (supply reliability)	Additional $4.67-$7.97 per month per household	$7.6-$12.9
Rollins et al. (1997)	CV	Canada	1,511	Estimate the WTP for a water conservation program, which can ensure adequate water service	Additional $26.00 per month on current water service charge	$39.6
Blamey (1999)	CE	Canberra, Australia	294	Estimate the WTP for possible water supply options (recycled water for outside use or drinking)	A$47 annual WTP for the provision of recycled water for outdoor use	$44.2
Koss and Khawaja (2001)	CV	California, USA	3,769	Estimate the WTP for improved water supply reliability (decreased water supply shortage)	$11.61 per month to avoid a 10% shortage once every 10 years; $16.92 per month to avoid a 50% water shortage occurring every 20 years	$17.8

Author	Method	Location	No. of completed surveys	Object	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
Whittington et al. (2002)	CV	Kathmandu, Nepal	1,500	Estimate households' demand for improved water services provided by a private operator (more water supply and higher water quality)	$14.31 per month for 500 litres improved water supply for households who have private connection; $11.67 per month and $3.19 per month for private and shared water connection	$4.4-$19.9
MacDonald et al. (2005)	CE	Adelaide, Australia	337	Estimate the WTP for improved continuity of water supply	A$1.10 to A$4.40 per year for decreased duration of water service interruptions; A$6.00 to A$15.40 per year for decreased frequency of interruptions in water services	$1.0-$14.2
Hensher et al. (2005)	CE	Canberra, Australia	211	Estimate the WTP for reduced interruptions of water supply and reduced number of wastewater overflows	Monthly interruptions A$9.58; two interruptions per year A$41.51; A$116 to reduce number of wastewater overflow to one time per year;	$8.8-$106.8

Author	Method	Location	No. of completed surveys	Object	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
Nam and Son (2005)	CV and CE	Ho Chi Minh City, Vietnam	120	Estimate the WTP for improved water quality and stronger pressure	108,000 VND per month from a piped water household for the proposed improved water service; 33,000 VND per month from non-piped households for a change to a medium water quality; 48,000 VND per month from non-piped households for strong water pressure	$2.1-$8.2
Willis et al. (2005)	CE	Yorkshire, England	1,000	Estimate the benefits to water company customers of changes across various water service factors	£0.03 for each reduction in the number of water samples that failed purity tests; £0.32 for each percentage increase in the security of supply; £0.78 per year for every 1,000 fewer cases of water discoloration; £2.27 per year for every 1,000 fewer supply interruptions	$0.1-$5.0
Fujita et al. (2005)	CV	Iquitos city, Peru	1,000	Estimate the WTP for water services and improved sanitation services	24.18 sol per month for water services by household who currently do not receive water service; 8.81 sol per month for households with water service for improved water availability and water pressure	$3.2-$8.8

Author	Method	Location	No. of completed surveys	Object	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
Casey et al. (2006)	CV	Brazil	1,479	Estimate the WTP of citizens for universal access to water services in their homes	$5.61 per month (accounted for 2% of a household's annual income)	$6.5
Genius and Tsagarakis (2006)	CV	City of Heraklion, Greece	294	Estimate the WTP of residents in urban areas to ensure a fully reliable water supply	€13.8 in addition to 3 month water bills to ensure a continuous (24 hour) water supply and stable tap water quality	$14.3
Hensher et al. (2006)	CE	Canberra, Australia	416	Estimate households' and businesses' WTP to avoid drought water restrictions	A$11.95 per year to reduce frequency of restrictions from once every 10 years to once every 20 years; A$3.98 per year to reduce water restriction from once every 20 years to once every 30 years; A$1,104 (23% of current water bill) by business respondents to avoid severe restrictions	$3.5-$10.5 for household; $968 for business
Tapsuwan et al. (2007)	CE	Perth, Australia	414	Estimate households' WTP to avoid outdoor water restrictions	22% more on households' water usage bills to be able to use sprinklers up to 3 days a week; 50% more on water bills to finance a new source of supply instead of enduring severe water restrictions	N/A

Author	Method	Location	No. of completed surveys	Object	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
Genius et al. (2008)	CV	Rethymno, Greece	306	Estimate residents' WTP to avoid water supply shortages and improved tap water quality	€10.64 for improved water quality and quantity (accounted for 17.67% of average water bills)	$14.47
Snowball et al. (2008)	CE	Eastern Cape, South Africa	71	Estimate WTP for improvement in water services (improved drinking water quality and reduced water supply interruptions)	15.72% in addition to water bills for a decrease in bacterial quality from slight risk to no risk; 0.12% and 0.13% increase in their water bills separately for every reduction of one household experiencing water discoloration or interrupted water supply	N/A
Vásquez et al. (2009)	CV	Parral, Mexico	398	Estimate households' WTP for safe and reliable drinking water	22.68 to 229.75 Mexican peso in addition to current water bills as the median household WTP to access for safe drinking water in the house	$2.3-$22.8

Author	Method	Location	No. of completed surveys	Object	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
MacDonald et al. (2010)	CE	Adelaide, Australia	337	Estimate WTP for improved reliability of household water services (reduced duration of water outage)	$0.15 to reduce the duration of an interruption by one hour; $4.05 to reduce the number of annual outages by one	$0.15-$3.9
Wang et al. (2010)	CV	Chongqing, China	1,478	Estimate WTP for water service improvement (improved reliability of water supply, water quality; water draining system and sewage water service)	2.5 to 3.3 yuan per ton on average for water usage per month (accounted for 1.5 to 2% of monthly income)	$0.39-$0.52
Polyzou et al. (2011)	CV	City of Mytilene, Greece	152	Estimate citizens' monetary valuation for the improvement of tap water quality	€10.38 every 2 months for the improvement of drinking water quality (€12.69 for citizens who always drink tap water and €9.43 for those who never drink tap water)	$13.1-$17.6

Author	Method	Location	No. of completed surveys	Object	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
Cooper et al. (2011)	CV	NSW and Vic Australia	472	Estimate consumers' WTP to avoid urban water restrictions	$6-117 per year as the median WTP	$6.5-$127
Akram and Olmstead (2011)	CV	Lahore, Pakistan	193	Estimate the WTP for improved piped water quality and reduced supply interruptions.	$7.5 to $9 per month for piped water supply that is clean and drinkable directly from the tap separately (about 3 to 4 times the average monthly water bill); $3 to $6 per month for improved consistency of piped water supply (eliminating supply interruptions and pressure drops)	$3.1-$9.2
Tarfasa and Brouwer (2011)	CE	Ethiopia	170	Estimate households' WTP for improved water supply services (increased water supply days and improved water quality)	$0.6 for one extra day water supply without water quality improvement; $1.3 for one extra day water supply and with water quality improvement; $0.8 and $1.5 individually for 2 extra days water supply, without and with quality improvement; $1.1 and $1.8 separately for 3 extra days water supply, without and with water quality improvement	$0.6-$1.8
Awad (2012)	CV	West Bank	525	Estimate WTP for improved reliability of water supply	NIS 31.4 per month for reliable water supplies (including both improved quality and quantity)	$8.1

Author	Method	Location	No. of completed surveys	Object	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
Behailu et al. (2012)	CV	Shebedino District, Southern Ethiopia	635	Estimate households' WTP for safe drinking water supply	3.65 Ethiopian Birr per month for safe drinking water supply (accounted for 2.36% of average monthly income)	$0.2

Note: CV refers to contingent valuation method; CE refers to choice experiments method; * unless otherwise indicated $ = $US

Table 2: Groundwater valuation surveys

Author	Method	Location	No. of surveys	Study	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
Edwards (1988)	CV	Cape Cod coast, Massachusetts, USA	585	Estimate households' maximum WTP to prevent uncertain nitrate contamination of Cape Cod's sole source aquifer	$5 million (per 1000 households for 30 years) when the probability of supply increase by 25%; About $25 million when the probability of supply increase to 1.0	$10 million-$50 million
Torell et al. (1990)	Market value differences	High Plains aquifer, USA	N/A	Assess the market value of water in-storage on the High Plains aquifer, using price difference between irrigated and dry land farm sales	$1.09 as the value of water per acre-foot in Oklahoma to $9.5 per acre-foot in New Mexico	$1.99-$17.3
Poe and Bishop (1992)	CV	Portage County, Wisconsin, USA	537	Estimate residents' WTP for groundwater protection program (prevent groundwater from agriculture contamination)	$269.3, $414.8 and $257.1 per year respectively as the WTP by ex-ante no-info group, ex-ante with-info group and ex-post group. The groups were divided by whether they received background information on nitrates in their own well water	$428.1-$690.7
Shultz and Lindsay (1990)	CV	Dover, New Hampshire, USA	346	Estimate WTP for a hypothetical groundwater quality protection plan (protect groundwater from future pollution)	$129 per year in extra property taxes to support the plan	$245.8

Author	Method	Location	No. of surveys	Study	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
Sun et al. (1992)	CV	Southwest Georgia, USA	660	Estimate households' WTP to eliminate the potential for groundwater contamination from agricultural chemicals	$641 per year for groundwater pollution abatement	$1,067.4
Powell et al. (1994)	CV	Massachusetts, New York, and Pennsylvania, USA	Not available	Estimate the value of increased groundwater supply protection and pollution prevention	$61.55 per year for groundwater supply protection	$96.99
Stevens et al. (1997)	CV	Massachusetts, USA	537	Value groundwater protection program alternatives (aquifer protection district, town-wide water treatment facility, private pollution control device, purchase of bottled water and doing nothing)	WTP for aquifer program was the highest among other alternatives and the mean WTP was $35, $340 and $243 separately, per year per household for the binary choice model, traditional ratings model, and ratings difference model	$50.7-$493.7
Stenger and Willinger (1998)	CV	Alsatian aquifer, Western Europe	817	Estimate the value of groundwater quality protection	150FF to 180FF per person per year to preserve the quality of groundwater	$36.2-$46.6

Author	Method	Location	No. of surveys	Study	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
White et al. (2001)	CV	Waimea Plains, Nelson, New Zealand	180	Estimate the value of the groundwater resource in terms of benefits for irrigation, commercial/industrial use and bulk water supply	The marginal value of water to irrigators is $240 to $300 per allocated cubic metre; the lower bound of WTP for household to a 20% reduction in groundwater extraction is $183 per household per year	$101.5-$166.5
Kerr et al. (2001)	CV	Christchurch, New Zealand	256	Estimate the WTP of meeting water needs by drawing and treating water from the Waimakariri River or from Ellesmere groundwater	$628-$640 to get more supply of water from the river; $527-$2,386 to get more supply of water from groundwater	$292.4-$1323.9
Hasler et al. (2005)	CV and CE	Denmark	600 for CE; 584 for CV	Estimate the value of groundwater protection	Using CE: 1,899DKK per year for naturally clean groundwater; 1,204DKK per year for water with very good conditions for plant and animal life; 912DKK per year for purified water using CE; Using CV: 711DKK and 529DKK for groundwater protection and purified water separately	$103.0-$370.6

Author	Method	Location	No. of surveys	Study	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
Aulong and Rinaudo (2008)	CV	Upper Rhine Valley aquifer, France	668	Estimate WTP for groundwater protection	€42.6 per year to restore drinking water quality; €77 per year to eliminate all traces of polluting substances	$57.92-$104.7
Martínez-Paz and Perni (2011)	Production function method and CV	Gavilan Aquifer, Spain	309	Estimate the total economic value of groundwater resources	0.381 €/m^3 as the value of groundwater for agriculture; 0.010 €/m^3 as the value of groundwater for recreational activities; 0.063 €/m^3 as the value of groundwater for environmental functions	0.01$/$m^3-0.5/m^3

Note: CV refers to contingent valuation method; CE refers to choice experiments method; * unless otherwise indicated $ = $US

Table 3 Wastewater valuation surveys

Author	Method	Location	No. of completed surveys	Study	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
Gillespie and Bennett (1999)	CV	Vaucluse, Sydney, Australia	306	Estimate environmental benefits from two sewage treatment proposals (a tunnel or a sewage treatment plant)	$137 as the median WTP for Vaucluse Area tunnel option; $76 as the median WTP for the sewage treatment plant option	$68.7-$124
Hoehn and Krieger (2000)	CV	Cairo, Egypt	903	Estimate benefits of water and wastewater service improvements	$7.77 per month for water connection project; $7.57 per month for wastewater connection project; $3.20 per month for improved reliability of the existing water services; $2.22 per month for wastewater network maintenance	$3.3-$11.6
Kontogianni et al. (2003)	CV	Thermaikos Bay, Greece	466	Examine residents' WTP to ensure the full operation of the wastewater treatment plant to improve water quality of Thermaikos Bay	€15.23 increase in the household four monthly water rates	$17.8
Genius et al. (2005)	CV	North-West Crete, Greece	326	Estimate WTP for wastewater treatment plant	€44 increase in quarterly water bills for wastewater treatment plant	$47.5

Author	Method	Location	No. of completed surveys	Study	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2012)
Tziakis et al. (2009)	CV	Municipality of Kissamos, northwest Crete, Greece	450	Estimate residents' WTP for a centralized wastewater treatment plant	€21.02 in addition to average quarterly water bills for wastewater treatment plant	$28.6
Birol and Das (2010)	CE	Chandernagore municipality, India	150	Estimate residents' WTP for improved capacity and technology of a sewage treatment plant	Rs100.32 per year in addition to municipal taxes to improved wastewater treatment plant quality	$2.3

Note: CV refers to contingent valuation method; CE refers to choice experiments method; * unless otherwise indicated $ = $US

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