CRC for
Water Sensitive Cities

Review of non-market values of water sensitive systems and practices: An update

Asha Gunawardena, Fan Zhang, James Fogarty and Sayed Iftekhar

Australian Government
Department of Industry, Innovation and Science

Business
Cooperative Research Centres Programme

Review of non-market values of water sensitive systems and practices: An update
Comprehensive economic evaluation framework (IRP2)
IRP2-3-2017

Authors
Asha Gunawardena[1,2], Fan Zhang[2], James Fogarty[1,2] and Sayed Iftekhar[1,2]

[1] CRC for Water Sensitive Cities

[2] Agricultural and Resource Economics, UWA School of Agriculture and Environment

Publisher

Cooperative Research Centre for Water Sensitive Cities
Level 1, 8 Scenic Blvd, Clayton Campus
Monash University
Clayton, VIC 3800

p. +61 3 9902 4985
e. admin@crcwsc.org.au
w. www.watersensitivecities.org.au

Date of publication: December 2017

An appropriate citation for this document is:

Gunawardena, A., Zhang, F., Fogarty, J., Iftekhar, M. S., (2017). Review of non-market values of water sensitive systems and practices: An update. Melbourne, Australia: Cooperative Research Centre for Water Sensitive Cities

Contents

Executive summary

Water sensitive systems and practices provide multiple indirect benefits such as environmental and ecological services. However, due to a lack of information on the monetized value of these services many of these services are not considered during investment decision making. The compilation and synthesis of existing non-market valuation estimates of the monetary value of water sensitive systems and practices, by making reference values readily available, can encourage the incorporation of indirect benefit value information in the investment decision process. Reference information on the monetary value of indirect benefits can also be used when conducting comprehensive post-hoc analysis of investments in water management projects.

This review presents the findings from an extensive survey of the literature on the key benefits and services delivered through the use of water sensitive systems and practices. The information is summarized in terms of major services: values associated with green infrastructure; ecological and environmental values of water; benefits of climate change mitigation; non-point source pollution reduction; flood hazard reduction; improved groundwater conditions; securing reliable water supply; and wastewater management. The review is primarily focused on non-market benefits, but in some cases estimates of market benefits (or tangible benefits captured through existing markets) and cost information are also presented.

In total, we have reviewed 345 studies; out of which 194 studies (56%) reported relevant non-market value information. A summary description of study attributes is presented in Table 1, where studies have been classified in terms of the method of analysis used and whether the study was Australian or not. Among the 194 studies with specific monetary non-market value information, around 23% were Australian. Overall, the most commonly studied topic was green infrastructure (30%), followed by water supply and pricing (21%). These two topic areas were also the two most common topic areas for Australian studies, although non-point source pollution has also been a relatively large focus in Australia.

Table 1: Summary of reviewed studies with non-market values

Theme	Benefits						Cost	Total	Australian	Total (%)	Australian (%)
	TC	HP	CV	CE	Meta	Other					
Green infrastructure	0	29	13	2	1	13		58	11	30	25
Amenity and recreational benefits		29	13	2	1	3		48	11	25	25
Air pollution and health						7		7	0	4	0
Energy savings						3		3	0	2	0
Ecological and environmental value of water	5	3	10	8	4	3		33	10	17	23
Water quality	5	0	7	5	4	0		21	3	11	7
Habitat conservation			2	1				3	0	2	0
Aesthetic value		1	1					2	0	1	0
Stormwater management		2		2		3		7	7	4	16
Climate change mitigation		1	1	1		7		10	1	5	2
Non-point source pollution			4	3			4	11	7	6	16
Flood hazard reduction		2			1		5	8	3	4	7
Recharge and improved groundwater quality			14	2		2	2	20	2	10	5
Water supply and pricing			25	12	1	3		41	7	21	16
Wastewater management			10	1		2		13	3	7	7
Total	**5**	**35**	**77**	**29**	**7**	**30**	**11**	**194**	**44**	**100**	**100**

Note: TC – Travel Cost, HP – Hedonic Pricing, CV – Contingent Valuation, Meta – Meta-analysis

The review found that for green infrastructure, the focus for research has been to estimate values for amenity benefits, and both stated preference and revealed preference methods have been used to estimate benefits. Reviewing the estimates in a systematic manner has revealed that estimates vary by location, the type of green infrastructure, distance to the asset, and estimation method. The review also found that economic valuation studies are rare for services such as mental and physical improvement, overall improvement of well-being, and improvement in liveability. This indicates a potential gap in economic assessment of the total benefits of green infrastructure.

In the existing literature there are some estimates of climate change mitigation benefits (such as urban heat island mitigation benefits, carbon sequestration and reduced carbon emission benefits) using direct measurements (such as value of carbon or capacity of trees to store carbons), but the literature is still developing. Some studies also use indirect measurement approaches, such as the loss in productivity due to extreme heat.

The interrelationship between people's preferences for different climate change mitigation options, and the benefits of green infrastructure does not appear to be well developed in the literature, as yet.

The existing non-market valuation studies related to water quality improvement are concentrated in the US. Generally, evaluations have been focused on water quality, water quantity, and its time and location distributions, with water quality the main focus area. Overall it remains the case that the ecological and environmental values of water are difficult to measure, with the ecosystem benefits provided to humans still not well understood.

Non-market valuation estimates on water quality improvement could be directly linked to the benefits of non-point source pollution reduction. However, the inclusion of non-market benefits in the formal analysis of various pollution control approaches is generally not seen in the literature. Rather than looking at the total value (or benefits) of different options, generally the abatement cost (based on life cycle cost analysis) of removing pollution from waterbodies/stormwater is estimated.

Natural cause related flooding is generally relatively localized and concentrated. The main impacts are also generally direct, which makes them easier to measure. For these reasons economic assessments of flood damage and different control options take a relatively standardized form, with heuristics like stage-damage relationships often used to estimate the benefits of different flood management options. The review did identify some estimates of the broader non-market benefits of different flood management options through improved stormwater management and it is possible to use these non-market benefits to evaluate different flood management options.

A substantial portion of the Australian economy depends on groundwater resources, and there are partial estimates of direct use values of different groundwater systems. The available estimates of direct uses reveal a high contribution of groundwater to the wellbeing of the community. However, groundwater management, information on non-market values is relatively limited. The subsequent generation of such information would help in understanding the total value of groundwater systems.

Studies on water supply and pricing provide a comprehensive assessment of people's willingness to pay to ensure a reliable supply of good quality water. The review did not consider the standard economic analyses of income and price elasticities, but rather had a focus on non-market value estimates. The non-market value estimates reveal that people's willingness to pay depends on not only the supply option considered, but also the baseline water supply service option, and the socio-economic conditions of customers.

Finally, the review considers studies focusing on wastewater management. The review found that people are willing to use recycled wastewater, but, mostly for outdoor and non-contact uses. Wastewater recycling projects can also provide other benefits, such as the: production of electricity and biogas; production of fertilizers; reduction of pollution load in downstream waterbodies; and reduction of pressure on existing infrastructure. The literature on these benefits is still evolving.

Glossary of selected terms

Choice Experiment (CE): A non-market valuation technique where willingness to pay is elicited by surveys in which people can choose between different bundles of goods with varying characteristics. The goods could be market or non-market goods (Meyer et al., 2014).

Contingent Valuation (CV) method: A non-market valuation technique where people are asked in surveys about their willingness to pay to avoid (or gain) a given decrement (or increment) of a particular non-market good, or about their willingness to accept its deterioration by receiving a certain amount of compensation (Meyer et al., 2014).

Control or prevention costs, averting behaviour: This method relies on the assumption that it is possible to quantify the economic value of externalities in terms of the avoidance costs of implementing actions that prevent the damage produced (Holguín-Veras et al., 2016).

Cost of Illness approach: This approach uses costs of health impacts (such as medical costs and lost wages due to illness) to estimate the value of a good or project (Meyer et al., 2014).

Damage (restoration) costs approach: This approach relies on quantifying the value of the impacts as the cost required to repair the damage, and restore things to their original condition (Holguín-Veras et al., 2016).

Hedonic pricing approach: This technique uses existing market price information to estimate the impact of a project or services. For example, by comparing the prices of similar houses in different areas of a city it is possible to estimate the capitalized amenity values of green infrastructures.

Life Satisfaction Analysis: Welfare estimations of public goods (health, environment) are estimated based on life satisfaction surveys (Meyer et al., 2014).

Production Function approach: This approach relies on estimating the contribution of an environmental good in producing a market good (Meyer et al., 2014).

Replacement Cost method: The value of an ecosystem good or service is estimated based on the costs of replacing that good or service (Meyer et al., 2014).

Stated preference (SP) techniques: Stated preference techniques use surveys to understand their preferences. Contingent Valuation and Choice Experiments are prominent examples of stated preference techniques (Holguín-Veras et al., 2016).

Travel Cost method: Recreational or environmental sites are valued by analysing observed travel time and expenditure of visitors (Meyer et al., 2014).

Willingness to accept (WTA): WTA is the amount that a decision maker is willing to accept to give up using a good or service, or to put up with a decrease in welfare (Holguín-Veras et al., 2016).

Willingness to pay (WTP): Willingness to pay is the amount of money that a decision maker is willing to part with to procure a good or service, or to achieve a higher level of welfare (Holguín-Veras et al., 2016).

Introduction

Urbanization is happening at a rapid rate across the world. Current urban areas cover less than 5% of the terrestrial surface. However, if the current trend in population growth continues, it has been predicted that by 2030 urban areas will triple in size; an increase of 1.2 million square kilometres over 2000 (Seto et al., 2012). Globally, more than 50% of the population already live in urban areas, and this proportion will increase substantially. In many developed countries the proportion of the population living in cities is already well about 50%. For example in Australia, the proportion of the population that lives in urban areas is above 80% of the population (Commonwealth of Australia, 2015).

Many Australian cities and towns are facing challenges from population growth and climate change. A growing population puts increasing pressure on the water supply system, as well as on wastewater treatment infrastructure. Supplying additional water is also an increasing challenge due to the rapid decline of some traditional water sources, such as groundwater from over-extraction, and the effects of climate change on dam supply. People's lifestyle choices are also evolving, and the demand for liveable environments with high amenity services is increasing. Adding to the complexity of the water management context, water authorities and utilities are also facing increasing pressure to improve the efficiency and cost-effectiveness of their investments (Sandhu and Wratten, 2013).

Water sensitive systems and practices treat water management in an integrated and systematic way, and as such, they are planning and design philosophies that can help address critical water management challenges. Specifically, the core concepts integrate the urban water cycle — including water supply, stormwater, groundwater, and wastewater management — into urban design (Brown and Farrelly, 2009).

The adoption of water sensitive systems and practices can provide tangible benefits or services, which are easily quantifiable, such as additional water supply; or intangible benefits, which are often difficult to quantify, for example the amenity benefit due to installing a rain garden. While the tangible benefits are easy to identify and incorporate into an economic analysis, intangible benefits are often difficult to include in economic assessments due to the absence of appropriate monetized values (Wong et al., 2013). In such situations, intangible benefits are often ignored in the formal investment decision framework.

To estimate monetized (non-market) values of intangible benefits and services economists have developed several specific valuation techniques. There are two main types of valuation technique: stated preference methods and revealed preference methods. Revealed preference methods use existing market price information to calculate the implied non-market values of goods and services. Stated preference methods use surveys to estimate people's preferences. A more detailed summary of the different techniques is been presented in the Appendix.

Researchers, using both stated preference and revealed preference methods, have conducted many studies to obtain non-market value estimates, and some studies have focussed on the non-market benefits generated from water sensitive systems and practices. The systematic compilation and review of the findings from these studies makes it possible to develop an understanding of the current state of knowledge in the area. The subsequent incorporation of this knowledge into the investment decision making process then enables better investment decisions to be made.

This review provides a summary of existing knowledge base. The review is based on a structured and comprehensive review of both the academic and grey literature. The review has been organised in terms of key services or benefits from water sensitive cities and practices:

- Non-market benefits from green infrastructure
- Ecological and environmental benefits of water
- Climate change mitigation options
- Pollution control and water quality improvement
- Groundwater management
- Water security and supply and
- Wastewater management

This review is an updated and revised version of a previously published CRC report: Zhang, F. and Fogarty, J. (2014). *Non-market Valuation of Water Sensitive Cities: Current Knowledge and Issues*, CRC for Water Sensitive Cities. The revision process involved adding new material on non-market benefits from green infrastructure and climate change based; adding recent publications to all studies and reporting value information in a common format.

Wherever feasible, information has been synthesised into table format. The summary tables allows the reader to quickly gain an overview of the literature. A key feature of the main summary table is that study values have been converted into a common metric: 2016 US$ (US$1 = AU$1.355, 2016, Reserve Bank Australia). The choice of US dollars was to ensure easier comparability across the international literature, and this standardisation helps readers to understand the magnitude and extent of non-market benefits in different contexts. However, under the summary tables the conversion rate to AU$ has been provided for interested readers. The values have not been adjusted to reflect income differences across countries and so it is necessary to excise care when making cross country comparisons. Where review of the summary information reveals an especially relevant study, we recommend readers consider to the characteristics and context of original study before using these values for any particular application, concluding benefit–cost analysis.

Green infrastructure

In many cities and towns open green space is rapidly shrinking. The loss of open green space is in turn resulting in the loss of ecosystem services (Vandermeulen et al., 2011). Empirical evidence suggests that urban green space / green infrastructure improves liveability (Badland et al., 2014). The pathways for green infrastructure to provide improved liveability are varied, and include: through providing increased amenity and recreational benefits (Brander and Koetse, 2011); heat mitigation benefits (Bowler et al., 2010, Norton et al., 2015); allowing for an increase in physical activities (Giles-Corti et al., 2005); reduced air pollution (Nowak et al., 2013); reduced noise pollution (Gidlöf-Gunnarsson and Öhrström, 2007); energy savings and lower carbon emissions (Derkzen et al., 2017, Pandit and Laband, 2010b); and improved resilience to natural disasters and enhanced community social capital (Hong and Guo, 2017). Urban green space also provides a wide range of ecosystem services, including: conservation benefits of urban biodiversity benefits (Hostetler et al., 2011, Hungate et al., 2017); and agri-ecological services such as water retention and water purification (Brander and Koetse, 2011).

This review focuses on the valuation of public open space or green infrastructure in urban settings, as public investment decisions are mostly made in this realm. The review considers valuation studies from across the globe and covers many different types of green infrastructure. Examples include urban parks, urban forests, and natural areas. Studies concerned with green spaces, including sports fields, under developed lands, agricultural lands on the urban fringe, and urban wetlands have also been reviewed. The available economic valuation estimates have been summarised in the following five dimensions: amenity; recreation; health benefits; energy savings; and improved productivity in workplaces. Table 2 provides a list of papers that provided estimates of the non-market values related to green infrastructure.

Amenity values

Revealed preference studies

The majority of studies on the value of green space use the hedonic pricing method to estimate amenity values. In hedonic pricing methods it is assumed that the price of residential properties reflect how people value different attributes of properties as well as positive or negative externalities due to surrounding land use. The importance of housing and land use related attributes are inferred using a regression analysis of property values on various attributes. Overall, the hedonic valuation studies show that availability of green space results in higher real estate prices, where the specific increase in value depends on the location, quality, functions, and size of the green space.

The effect of spatial location and proximity to green space on property prices have been evaluated in several studies, and Brander and Koetse (2011) conducted a meta-analysis of 12 hedonic studies from the US and concluded that 10 m decrease in distance to the green open space results in 1% increase in the average house price in 2003.

Anderson and West (2006) carried out a study in St. Paul, US (based on dataset from 1997) and found that halving the distance to the nearest park would increase the sale price of an average house by $142 per year.

Cho et al. (2008) examined property sale prices in the City of Knoxville, Tennessee in 2000, and found that moving 1 km closer to the evergreen forest increased the price of an average house by $692. However, the same study concluded that moving 100 m closer to a deciduous forest patch decreased the price of the average house price by $589.

Nicholls and Crompton (2005) found that in Texas, US properties located directly adjacent to a greenbelt, on average were associated with an increase in value of $44,332.

Mansfield et al. (2005) conducted a study of properties in North Carolina and that found adjacency to a private forest block increased the average house price by more than $8,000.

Votsis (2017) investigated apartment transactions in Helsinki, Finland using a dataset from 2000-2011. The study found that on a multi-year average, the effect of a 100 m increase in distance to a forest resulted a decrease of 3.7% in price/m^2 in the city centre but that the effect was zero at 6 km from the CBD.

Morancho (2003) conducted a study in France based on property sales in the City of Castellón using data from 2001. The study found that for every 100 m further away from green area house prices fall by €1,800.

Using property sales data in the Netherlands from 1989-1992, Luttik (2000) found that properties located within walking distance to a park were associated with a house price premium of 6%.

In contrast to the findings of most studies, Jun and Kim (2017) found that in that in Seoul, Korea moving 1 km closer the nearest green belt would reduce apartment rents by 3.8% - 3.9%.

Using data for 2005-2006 Jim and Chen (2010) found that in Hong Kong having a neighbourhood park near the apartment building was associated with an apartment price increase of 14.9%.

Sander et al. (2010) examined property sales in Minnesota, US in 2005 and found that a 10% increase in tree cover within 100 m of a property added $1,371 to the average property price but that at a distance of 250 m a 10% increase in tree cover added only $836 in value.

Donovan and Butry (2010) studied the value of trees in Portland, Oregon, US using property sales data for 2007 and found that, on average, street trees add $8,870 to the sale price of a home. In a related study, using 2009 and 2010 sales data for Portland, Donovan and Butry (2011) find that an additional tree on a house's lot increases the monthly rental price by $5.62; and that a tree in adjacent public space increases the monthly rental price by $21.

Netusil et al. (2014) examined the proximity, abundance and characteristics of green street facilities using in Portland using data for 2005-2007. The study found that a 10% point increase in tree canopy at the closest green street facility was associated with an increase a property's sale price of $18,707.

Using property transaction records from 2003-2004 in the city of Los Angeles, US, Saphores and Li (2012) found that the benefit of adding one generic tree with a 16 m^2 canopy cover over the impervious area of a house was an increase of property value by $204.

Using data on the sale price of single-family houses sold between 1997 and 2006 in Virginia, US, Poudyal et al. (2009a) found that 10% increase in square footage of an urban park in the neighbourhood increased house values by 0.03%. Poudyal et al. (2009b), also investigated the spatial pattern and appearance of the green space and found that residents preferred open spaces in few larger plots to many smaller pieces that are scattered throughout the neighbourhood.

Rossetti (2013) use property sales data from 2000 to 2010 across Australian cities to measure the impact of proximity to green infrastructure proxied via a postcode level measure of green vegetation. The study found that a one standard deviation increase in green infrastructure is associated with an increase in house prices of between $A32,000 and $A58,000.

Plant et al. (2017) is a study of 52 residential suburbs in Brisbane, using 2010 property sales price data. The study found that a 1% increase in tree cover along the foot path, within 100 m of a property, results in an increase in property values of between 0.08% and 0.1%.

In a study of house prices in Perth, Pandit et al. (2014) found that 10% increase in tree canopy cover on the adjacent public space was associated with an increase in property prices premium of about AU$14,500 in 2009.

Several hedonic studies have looked at how much value residents place on having a scenic view of open urban space. Specifically, Jim and Chen (2010) found having a view of a neighbourhood park increases property prices by 1.9%; Jim and Chen (2006b) found that in Guangzhou, China having a scenic view of a green space increased property values by 7.1%; and Luttik (2000) find that a view of open green space in the Netherland was associated with a 9% increase in house prices.

In AECOM (2017), the impact of a 10% increase in canopy cover in the Sydney suburbs of Annandale, Blacktown, and Willough is investigated. Using property sales data for the periods 2012 to 2016, the study found that a 10% increase in canopy is associated with an increase property prices of around $50,000.

Stated preference studies

The other common approaches to estimate values that people place on the amenity provided by green space are contingent valuation (CV) studies and choice experiments (CE).

A meta-analysis of the contingent valuation literature is presented in Brander and Koetse (2011). The study considers 20 contingent valuation studies from several countries and estimated the value of open space per hectare per year in 2003 as US$13,210.

For example, Andrews et al. (2017), reports the results of s survey of 386 households in Norwich, UK, in 2009 that found the WTP of residents to have a park in the city centre was £23.14 per household, and that the WTP for of residents for a suburban park was £19.11 per household.

Mell et al. (2013) investigates the value of green infrastructure in Manchester using a contingent valuation survey. The study collected data from 512 respondents and found that residents were WTP £1.88 more per month to improve their local environment with green investment. Commuters and local workers in the area who were not residents were found to be WTP slightly less than residents: £1.60 to 1.65 per month.

Chen and Jim (2008) conducted a contingent valuation study in Zhuhai, China. The study relies on data from 598 respondents and was conducted in 2006. The study estimated that the mean WTP for a greening the local environment project was 161.84 Renminbi per household per year.

del Saz Salazar and Menendez (2007) carried out a contingent valuation survey in Valencia in 2001, Spain to investigate WTP for a new urban park. The study findings suggest that individuals living relatively close to the site were willing to pay 11,238 -14,497 Pesetas (€67.48-€69.02) while the WTP for those who live away from the site ranged from 7,830 to 8,571 Pesetas (€47.01- €51.45).

The willingness to pay for a new urban park was also investigated in Latinopoulos et al. (2016). The study location was Thessaloniki, Greece, and the results rely on a survey of 600 inhabitants in 2013. The study found that households, on average, would be willing to pay around €2.0 to € 2.25 per month in terms of a “green tax” for a new urban park.

Tu et al. (2016) used the choice experiment method to estimate the value of peri-urban forests based on data from 180 respondents in Nancy, France in 2013. The WTP of home owners who did not have a private garden was found to be € 34.84/m^2 (that was 2.7% of their current average house price) while for those with a private garden WTP was found to be € 16.42/m^2 (1.2% of their current average house price). This result is interpreted as evidence of a substitution effect between public and private open space.

del Saz-Salazar and Rausell-Köster (2008) consider the social benefits of a large existing park in Valencia, Spain. The study is based on 1,480 face-to-face interviews conducted in 2005 and found that on average people were willing to pay €7.60/person for the social benefits created by the park.

Jim and Chen (2006c), estimated the recreational value of existing green spaces in Guangzhou, China by conducting contingent valuation survey of 340 people. The study reported that the WTP of individuals for recreational and amenity benefits was $2.1/person/month.

Use the contingent valuation method, Pepper et al. (2005) estimated the mean WTP for the preservation of Hartfield Park bushland in the Perth Metropolitan area, Western Australia as AU$21.60 per person per annum.

In Dumenu (2013) the contingent valuation method is used to estimate the economic value of preserving an urban forest in Ghana. The study relies on a survey of 200 residents and found the mean WTP for the preservation of urban forest was US$22.55/year.

Lo and Jim (2010) use the contingent valuation method to estimate the value the conservation of urban green spaces in Hong Kong. The study relies on a survey of 495 urban residents in 2008. The findings suggest that WTP to prevent a 20% loss of urban green space was US$9.90 per household for five years.

Vesely (2007) uses the contingent valuation method to estimate WTP for an avoidance of a 20% reduction in local urban tree estates. The survey was conducted across 15 cities in New Zealand in 2003, and the sample size was 344 households. The study found that households, on average, would be willing to pay NZ$184 annually to prevent the loss.

The benefits of reclaiming urban quarries in the centre of Athens, Greece were assessed in Damigos and Kaliampakos (2003). The study results are based on a survey of 200 households and the mean WTP reforestation was €29.44 to €30.75; backfilling and reforestation €45.88 to €49.47.

Other methods

There are also studies that estimate implicit prices for the relationship between environment and wellbeing of people using self-reported life satisfaction or subjective wellbeing measures, for example Smyth et al. (2008) and Welsch (2002).

In terms of studies that report dollar value metrics, in Ambrey and Fleming (2014) self-reported life satisfaction data from the Household Income and Labour Dynamics in Australia (HILDA) survey in 2005 is used to estimate the willingness to pay for urban green space by residents in Australian capital cities. The study found households in Australian capital cities are willing to pay $1,172 per annum for one% increase in open public space in their local area. On average a 1% increase public open space was equivalent to an increase of 143 square metres.

The State Government of Victoria has undertaken a study to value Victorian parks (The State of Victoria Department of Environment, 2015). The study used best practice environmental accounting to quantify the benefits that parks and their ecosystems provide. In the study the total amenity value provided by parks for Victorian residents was estimated at between $21 million and $28 million per year.

Recreational values

Mahmoudi et al. (2013) is a hedonic study that was conducted in the Adelaide metropolitan area using property sales data from 2005 to 2008. The study found that being 1 m closer to a Golf course, green space sport facilities and the coast increased property prices by $0.54, $1.58 and $4.99 respectively

Pandit et al. (2013) used the hedonic price method to estimate the value of proximity to different types of recreational asset in Perth, Western Australia. The study found that an increase of 1 m in distance to a larger park, where bushwalking is possible, reduced the property values by $9.60; and an increase of 1 m in distance to a sports reserve decreased property values, on average, by $29.59.

Using contingent valuation method, Bernath and Roschewitz (2008) estimated recreational benefits of the Zurich city forests in 2004. The study was based on analysing data from 1,500 residents of Zurich and found that visitors' willingness to pay for an annual forest visitor permit was between $64 and $91.

Health benefits due to reduced air pollution

Urban trees can remove air pollution by the interception of particulate matter on plant surfaces and the absorption of gaseous pollutants through the leaf stomata. A number of studies have estimated air pollution removal benefits of urban trees and shrubs in the United States. For example, Nowak et al. (2006) estimated pollution (O_3, PM_{10}, NO_2, SO_2, CO) removal from urban green space across the 48 contiguous states of the US as 711,000 metric tons using pollution concentration data from in 1994. The value of this pollution removal was estimated to be worth $3.8 billion.

Nowak et al. (2013) is a study conducted in 10 US cities in 2010 where $PM_{2.5}$ concentrations and human health are considered. The study estimated the total amount of $PM_{2.5}$ removed annually by trees in cities of different size, with the annual value of pollutants removed ranging from $1.1 million in Syracuse to $60.1 million in New York City. The average health benefit value per hectare of tree cover was estimated about $1,600, but varied with city size, population density, and pollution load. The highest value reported was for New York City, where the health benefit value was estimated at $3800 per hectare of tree cover.

In Nowak et al. (2014) computer simulations using the U.S. EPA's BenMAP program are used to estimate the value of the pollutant load removed by trees and forests in the US. Using 2010 as the calibration year, the study found that trees and forests in the 48 contiguous states of the US removed 17.4 million tonnes of air pollution, annually, and the annual value of the pollution removed was US$ 6.8 billion.

Tallis et al. (2011) examined tree survey data and annual maps of the PM_{10} distribution in 2006 for the Greater London Authority, UK using Urban Forest Effect Model. The study estimated that the annual removal of atmospheric particulate pollution due to tree cover was between 852 and 2,121 tonnes.

Jim and Chen (2008) conducted a study in Guangzhou, China in 2000. Using the variation in the urban land uses in the city the study found that annual benefits gained due to removal of air pollutants from urban green space is about RMB90200.

Based on a field survey conducted in 2002, Yang et al. (2005) estimate that in Beijing, China the air pollution removal by trees in the central part of Beijing was 1261.4 tonnes, and the carbon dioxide (CO_2) stored in biomass form by the urban forest was about 0.2 million tonnes.

Yang et al. (2008) examined 170 green roofs in Chicago, US using a dry decomposition model and found that the total air pollutants removed by 19.8 ha of green roofs in one year was about 1,675 kg. The reference year for the study was 2002.

Improvements in physical and mental health

A growing body of literature has emerged on the health benefits of having contact with nature. Much of this literature has focused on urban green spaces as a readily available type of nearby nature with a high potential for health and wellbeing benefits. Many studies across the globe confirm that natural open spaces play an important role in facilitating physical activities and helping to address sedentary behaviours (Barton et al., 2009, Bedimo-Rung et al., 2005, Coombes et al., 2010, Giles-Corti et al., 2005, Hillsdon et al., 2006, Lee et al., 2015, Lee and Maheswaran, 2011, Tzoulas et al., 2007). Most studies are qualitative studies which tried to establish the relationship or develop a conceptual framework between green space and health.

A few studies examined the link between access to neighbourhood green space and mental health. For example, Alcock et al. (2014) analysed British Household Panel Survey data combined with mental health data from 1992 to 2008 and found that individuals who moved to greener areas had significantly better mental health in all three post move years.

A study carried out in Wisconsin also found that higher levels of neighbourhood green space were associated with lower level of depression among the residents (Beyer et al., 2014). A similar study undertaken in Perth, Western Australia from a cross sectional survey of residents in 2003 and 2005 concluded that residents in neighbourhoods with high quality public open space had higher odds of low psychosocial distress than residents of neighbourhoods with low quality public open space (Francis et al., 2012).

Zhang et al. (2015c) also reported a positive relationship between attachment to local green space and better self-reported mental health in the neighbourhood in a medium sized city in Netherlands.

Lafortezza et al. (2009) investigated the perceived wellbeing of residents on the use of green space during heat stress periods in Italy and the UK. The study found that longer and more frequent visits to green spaces could generate significant improvements in the perceived wellbeing of users.

Sugiyama et al. (2008) examined the link between green space and both physical and mental health in Adelaide, South Australia. The study found that those who perceived their neighbourhood as highly green had, respectively, a 1.4 and 1.6 times higher chance of having better physical and mental health compared with those who reported living in a neighbourhood with the lowest level of perceived greenness.

A study carried out in Portland, Oregon using data from residents' birth certificates and tax records found that a 10% increase in tree-canopy cover within 50 m of a house reduced the number of low weight births (Donovan et al., 2011).

Konijnendijk et al. (2013) is a review of the relationship between urban parks and health outcomes and this review suggests that there is sufficient evidence to view parks as promoting health indirectly, particularly through increased physical activity, but that further research is required to establish a complete picture.

Although the relationship between green space and mental and physical health has been well established, it can be seen from the above discussion that remains a lack of estimates on the economic values of the benefits.

Energy saved

Studies have shown that having urban trees in the neighbourhood reduces electricity consumption, especially during summer time due to the shading and cooling effect provided by trees. For example, Donovan and Butry (2009) estimated the effect of shade trees on the summertime electricity savings of 460 single-family homes in Sacramento, California. The study results show that trees located on the west and south sides of a house (in the Northern Hemisphere) reduced summertime electricity use by 185 kWh, which, on average is equivalent to a 5.2% reduction is electricity use.

Pandit and Laband (2010a) examined the effects of trees on electricity use in Auburn, Alabama. The study found that every 10% of shade coverage, on average, reduced electricity consumption by 1.29 kW h/day. For a house with mean shade coverage of 19.3% during the summer months, dense shade reduces daily electricity consumption by 9.3%.

McPherson and Simpson (2003) is a study that was carried out for California. Using tree canopy cover data from aerial photographs the study simulated the energy savings for buildings from existing trees and new plantings. Existing trees were estimated to decrease annual air conditioning energy use by 2.5% with a wholesale value of this energy saving of $485.8 million, in 2010. The total peak load reduction by existing trees was estimated to save utilities $778.5 million annually. The estimated saving was $4.39 per tree.

Donovan and Butry (2009), reported that a London plane tree, planted on the west side of a house, can reduce carbon emissions from summertime electricity use by an average of 31%.

Green urban infrastructures also provide climate change and mitigation benefits by providing thermal comfort (Yu and Hien, 2006), storing carbon (Davies et al., 2011, Escobedo et al., 2010) and balancing water flows (Demuzere et al., 2014). A summary of relevant studies in this area is presented in the climate change section.

Improved productivity in workplaces

It has been suggested that green buildings provide economic and organizational benefits for business which in turn increase workplace productivity (Heerwagen, 2000, Miller et al., 2009, Clements-Croome, 2015). Two case studies were conducted in the area of Lansing, Michigan, with a retrospective–prospective cohort design to evaluate the effects of moves to green buildings on perceived employee outcomes. The study found that improved indoor environmental quality contributed to reductions in perceived absenteeism and work hours affected by asthma, respiratory allergies, depression, and stress and to self-reported improvements in productivity. These preliminary findings indicate that green buildings may positively affect public health (Singh et al., 2010). In Ries et al. (2006) the benefits of a green building design located near Pittsburgh, Pennsylvania were estimated. The study found that in the new facility manufacturing productivity increased by about 25%; energy usage decreased by about 30% on a square foot basis; and absenteeism fell. The study does however note that in addition to the new green building there are many factors that could impact productivity, such as the new work layout, new production equipment, and an employee psychology effect due to the new facility.

Table 2: Urban green space valuation studies

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Economic measures	Adjusted WTP value (value in $US2016)*
Amenity and recreational values: hedonic studies						
Anderson and West (2006)	HD	St. Paul metropolitan area, US	24,862 property sales (1997)	Halving the distance to nearest special park (Mean distance to the special park is 2265.40 m)	Increases the sales price of an average home by $142 per year (0.1%)	$212.34
Brander and Koetse (2011)	Meta-analysis of Hedonic studies	US	12 hedonic pricing studies (in 2003)	10 m decrease in distance to open space	0.1% increase in house price	
Cho et al. (2008)	HD	City of Knoxville, Tennessee, US	9571 house sales (2000)	(i.) At initial distance of 1 km, moving 100 m closer to an evergreen forest (ii.) Moving 100 m closer to a deciduous forest patch (iii.) An additional patch per hectare of forest in a neighbourhood (iv.) An additional meter of edge per hectare of forest (v.) An additional ha in average forest patch size in the neighbourhood	(i.) Increases the average house price by $692 in 2000 (evaluated at the mean house price of $117,787) (ii.) Decreases the average house price by $589 in 2000 (iii.) Decreases the price of a house by $62 (iv.) Increases the housing price by $35 (v.) Decreases the housing price by $1,178	(i.) $964.49 (ii.) $820.93 (iii.) $86.41 (iv.) $48.78 (v.) $1,641.86
Donovan and Butry (2010)	HD	Portland, Oregon	2608 houses (in 2007)	On average, street trees	Add $8,870 to sales price	$10,269.47
Donovan and Butry (2011)	HD	Portland, Oregon	985 rental prices (2009-2010)	An additional tree on a house's lot A tree in the public right of way	Increased monthly rent by $5.62 Increased rent by $21.00	$6.19 $23.11
Jim and Chen (2006a)	HD	Guangzhou, China	652 dwelling units (2003–2004)	View of green spaces Proximity to water bodies	Increased house price by 7.1% Increased house price by 13.2%	
Jim and Chen (2009)	HD	Hong Kong: harbor and mountain	1474 transactions in 2005 and 2006	(i.) A broad harbor view (ii.) A confined harbor view (iii.) A broad mountain view	(i.) Increased the value of an apartment by 2.97%, equivalent to $15,173 (ii.) Increased price by 2.18% or $11,137	(i.) $18,063.62 (ii.) $13,258.72

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Economic measures	Adjusted WTP value (value in $US2016)*
					(iii.) Decreased apartment price by 6.7%	
Jim and Chen (2010)	HD	High-rise residential units in Hong Kong.	1471 transactions in 2005 and 2006	Neighbourhood parks	Lift price by 16.88%, including 14.93% for availability and 1.95% for view	
Jun and Kim (2017)	HD	Seoul, Korea	3262 transactions with in 5 km of the greenbelt (2010)	One unit (1 km) decrease in the distance to the nearest greenbelt	Decreases the apartment rents by 3.83-3.95%: $34 drop in monthly rent	$37.42
Luttik (2000)	HD	8 Towns in Netherlands	3000 houses 1989-1992	In Apeldoorn, (i.) Walking distance to a park (400 m) (ii.) View of the park In Leiden (iii.) Walking distance to a park (400 m) (iv.) View of open space within 3-5 km of attractive landscape	(i.) A premium of 6% of the house price (ii.) A premium of 8% of the house price (iii.) A premium of 9% of the house price (iv.) A premium of 7% of the house price	
Mahmoudi et al. (2013)	HD	Adelaide metropolitan area	40923 properties (2005-2008)	(i.) 1 m closer to golf course (ii.) 1 m closer to greenspace sport facilities (iii.) 1 m closer to the coast	(i.) Property price increases by $0.54 (ii.) Property price increases by $1.58 (iii.) Property price increases by $4.99	(i.) $0.60 (ii.) $1.76 (iii.) $5.56
Morancho (2003)	HD	City of Castellón, France	810 houses (2001)	Every 100 m further away from a green area	A drop of €1800 in housing price	$2,065.51
Mansfield et al. (2005)	HD	Research Triangle region of North Carolina	11206 observations (1996 and 1998)	Adjacency to a private forest block	Increased house price by more than $8,000	$11,779.48
Netusil et al. (2014)	HD	Portland, Oregon	29,644 transactions (2005-2007)	(i.) Each additional dam (ii.) An increase in distance of 1 ft away from the nearest green street facility	(i.) Increased a property's sale price by 0.60% (ii.) Increases a property's sale price by $0.30 of which $0.20 is a direct effect and $0.10 is an indirect effect	(ii) $0.35 , $0.23, $0.12

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Economic measures	Adjusted WTP value (value in $US2016)*
				(iii.) A 10% point increase in tree canopy at the closest green street facility	(iii.) Increases a property's sale price by $18,707 of which $12,590 is a direct effect and $6,117 is an indirect effect	(iii) $21,658.52 - $14,576.40 – $7,012.82
Nicholls and Crompton (2005)	HD	Barton Creek, Austin, Texas Travis	224 properties (999-2001) 236 properties (1999-2001)	Directly adjacent to the Barton greenbelt Directly adjacent to the greenbelt	$44,332 increase in property value representing 12.2% average value of adjacent homes $14,777 increase in property value representing 5.7% average value of adjacent homes in Travis	$60,079 $20,025.88
Poudyal et al. (2009a)	HD	City of Roanoke, Virginia.	11,125 houses (1997-2006)	10% increase in square footage of the urban park in the neighbourhood (Mean house price is $95,133.99) 100 ft^2 increase in the size of the park	Increased the real sales price of the house by 0.03% $0.79 increase in price of nearby houses	$0.88
Poudyal et al. (2009b)	HD	City of Roanoke, Virginia.	A total of 11,125 single-family houses were sold (1997- 2006)	Having a variety of open spaces in the neighbourhood	Increased house prices	
Plant et al. (2017)	HD	52 Brisbane residential suburbs	2774 houses (in 2010)	Marginal implicit price for a 1% increase in footpath tree cover within 100 m	$US 312–393 (median house value is 530000) representing 0.082-0.103% premium of property price	$343.41 - $432.56
Pandit et al. (2014)	HD	central part of the Perth metropolitan area in WA	5606 houses (2009)	10% increase in tree canopy cover on the adjacent public space	Property price premium of about AU$14,500	$13,188.17
Pandit et al. (2013)	HD	23 northern suburbs of the Perth metropolitan region in WA	2149 properties (2006)	(i.) A broad leaved tree on street verge (public place) in 2006 (ii.) 1 m distance to a larger park (bushwalking) (iii.) 1 m distance to a sports reserve	(i.) Increases median property price of a house by AU$ 16,889 (4.27%). (median house price is 395,000) (ii.) Reduces the property value by $9.60 (median house price is 395,000) (iii.) Decreases the property value by $29.59	(i.) $15,186.21 (ii.) $8.63 (iii.) $26.61

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Economic measures	Adjusted WTP value (value in $US2016)*
Rossetti (2013)	HD	Australia	2,531,803 observations of property transactions (2000-2010)	For every house in a postcode that gains green infrastructure equivalent to 1 standard deviation change in enhanced vegetation index	$32,000 - $58,000 per property	$34,195.47 – $61,979.30
Sander et al. (2010)	HD	Ramsey and Dakota Counties, east central Minnesota, USA	9992 property sales (2005)	A 10% increase in tree cover within 100 m) A 10% increase in tree cover within 250 m)	Increases average home sale price by $1,371 (0.48%) Increases sale price by $836 (0.29%)	$1,684.84 $1,027.37
Sander and Haight (2012)	HD	Dakota County, Minnesota, USA	5094 single-family residential properties (2005)	(i.) Marginal implicit price of a 100 m decrease in distance to a park (evaluated at the mean home sale price of $319,073) from an initial distance of 1 km (ii.) A 1 ha increase in the area of lawn from the mean value (2584 m^2) in a home's view shed (iii.) 10% increase in tree cover within each of these four neighbourhoods from their mean values (evaluated at the mean home sale price)	(i.) $13.16 (0.040%) (ii.) Corresponded to a sale price increase of $1,742 (0.55%) (iii.) Increased house price by $1,853 (0.581%), $1,030 (0.323%), $1,947 (0.610%), and $1,102 (0.345%), respectively	(i.) $16.17 (ii.) $2,140.77 (iii.) $2,277.18, $1,265.78 $2,392.70, $1,354.26
Saphores and Li (2012)	HD	Los Angeles, CA	20,660 transactions (2003-2004)	Median benefit of adding one generic tree with a 16 m^2 canopy cover	Would increase its value by $204	$259.19
Votsis (2017)	HD	Helsinki, Finland	44.300 transactions (2000-2011)	On a multiyear average, a 100 m increase of distance to a forest	Decreases 3.7% of price/m^2 at 0 km from the CBD, which gradually drops to zero at 6 km from the CBD	
Xiao et al. (2016)	HD	Shangha, China	4188 housing transactions (2007-2009)	(i.) For each additional unit of the community green space ratio (ii.) One unit of additional public green space (iii.) For every km nearer to a city park (iv.) For every km away from public green space	(i.) Adds 8.7% to the property sale price (ii.) Has zero value for home buyers. (iii.) A premium of 2.6% (iv.) Home buyers pay extra 4.5% of house price	

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Economic measures	Adjusted WTP value (value in $US2016)*
Amenity and recreational values: life satisfaction approach						
Ambrey and Fleming (2014)	LS (Life satisfacton approach)	Australian capital cities	Household Income and Labour Dynamics in Australia (HILDA) survey. (2005)	Average implicit willingness to pay for a 1% increase in public greenspace A one standard deviation (12.49%) increase in public green space	$1,172 per household per year Approximately $12,800 per year	$940.14 $10,267.69
Amenity and recreational values: contingent valuation(cv) and choice experiments						
Andrews et al. (2017)	CV	Norwich, UK	386 completed surveys (2009)	Willingness to pay to have a park in the City Centre Have a Suburban park	£23.14 per household £19.11 per household	$42.79 $35.34
Bernath and Roschewitz (2008)	CV	Zurich city forests	1500 residents of Zurich (2004)	Visitors' willingness to pay for an annual forest visitor permit (initial bid) visitors' willingness to pay for an annual forest visitor permit (revised bid)	$64 $91	$81.32 $115.62
Brander and Koetse (2011)	Meta-analysis of CV studies	Several countries including US, UK. Canada, Australia, China and Finland	20 contingent valuation studies on urban and peri-urban open space (2003)	WTP per ha of green space per year in 2003	Mean value of $1,500 per ha per year	$1,956.58
Chen and Jim (2008)	CV	Zhuhai city in south China	598 respondents (2006)	Mean WTP for leisure value from urban green park Aggregate leisure value	RMB161.84 per household per year RMB12.3 million per year	$24.02 $1.82 million
Damigos and Kaliampakos (2003)	CV	Galatsi Municipality , Athens, Greece	200 households 1998 and 1999	Mean WTP pay for rehabilitation of an abandoned quarry site in the city under following alternatives (i.) Reforestation (ii.) Backfilling and reforestation (iii.) Partial backfilling, reforestation and new land uses	(i.) €30.75 (parametric mean €29.44) (ii.) €49.47 (parametric mean €45.88) (iii.) €58.20 (parametric mean €56.44)	(i.) $38.78 ($37.13) (ii.) $62.39 ($57.86) (iii.) $73.40 ($71.18)

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Economic measures	Adjusted WTP value (value in $US2016)*
del Saz Salazar and Menendez (2007)	CV	Valencia (Spain)	900 randomly chosen inhabitants (2001)	2. Provision of a new urban park (i.) Mean WTP for affected (long-term residents) (ii.) Mean WTP for the less affected (short-term residents)	(i.) 14,497 pesetas per person (parametric) 11,238 pesetas per person (non- parametric) (ii.) 8,571 pesetas per person(parametric) 7,830 pesetas per person (non- parametric)	(i.) $79.27 ($77.48) (ii.) $59.09 ($53.98)
del Saz-Salazar and Rausell-Köster (2008)	CV	City of Valencia (Spain)	1480 face-to-face interviews (2005)	Mean WTP for using urban park	€7.60 per year	$7.57
Dumenu (2013)	CV	Ghana	A total of 200 respondents	Overall mean WTP for preservation of urban forest	$22.55 per year	$23.57
Jim and Chen (2006c)	CV	Guangzhou, China	340 respondents (2003)	WTP for recreation in urban greenspace	$2.1 per person per month	$2.74
Latinopoulos et al. (2016)	CV	Thessaloniki, Greece	600 inhabitants 2013	Mean WTP to have a new park	Around €4.0 to € 4.5 as a bi-monthly "green tax" to the municipal authority	$5.35 - $6.02
Lo and Jim (2010)	CV	City of Hong Kong	A total of 495 urban residents from different neighbourhoods (2008)	WTP to recover a possible loss of urban green spaces area by 20%	Monthly average payment of HK$77.43 (approx. $9.9) per household for five years	$11.04
Mell et al. (2013)	CV	Manchester, UK	512 respondents (2011)	WTP by residents for investment in green infrastructure WTP by commuters and employees	£1.88 more per month £1.60–1.65 per month	$3.24 $2.75 - $2.84
Pepper et al. (2005)	CV	Perth, Western Australia	1000 questionnaires (54% responded (2001)	Mean WTP for the preservation of the bushland (Hartfield Park bushland)	AU$21.60 per person per annum	$14.79
Tu et al. (2016)	CE	Nancy	180 respondents (2013)	(i.) MWTP for living 100 m closer to a green park by middle- income home owners who do not have a private garden (ii.) MWTP for living 100 m closer to a park by middle-income homeowners who have a private garden	(i.) MWTP was 2.7% of their current house's price (€ 34.84/m^2) (ii.) MWTP was 1.2% of their current house's price (€ 16.42/m^2) on average (iii.) WTP was 1.4% of their actual rent (€ 0.12/month/m^2)	$46.62 $21.97 $0.16

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Economic measures	Adjusted WTP value (value in $US2016)*
				(iii.) MWTP for living 100 m closer to a green park by tenants who do not have a private garden (iv.) WTP of average respondents to have a scenic view of green spaces outside their window	(iv.) 9.9% more of their current average house price	
Vesely (2007)	CV and CE	15 cities in Aotearoa New Zealand	344 respondents (2003)	On average, households would be WTP for the avoidance of a 20% reduction in their local urban tree estate,	NZ$184 annually covering a period of 3 years	$138.57
Air pollution removal by green space and health benefits						
Jim and Chen (2008)	Previous study estimates	Urban trees in Guangzhou (China)	Different land uses were acquired from different monitoring systems (2000)	An annual removal of SO_2, NO_2 and total suspended particulates Benefits gained due to removal of air pollutants	About 312.03 mg RMB90.19 thousand ($1.00 = RMB8.26)	 $1038 thousand
Nowak et al. (2006)	Estimates from previous studies	United States	Pollution concentration data from across the coterminous US (1994)	Total annual air pollution removal (O_3, PM_{10}, NO_2, SO_2, CO) estimated	711,000 metric tons ($3.8 billion value)	$6.15 billion
Nowak et al. (2013)	Environmental Benefits Mapping and Analysis Program (BenMAP) model	10 U.S. cities	Field data on trees were measured within randomly selected 0.04 ha plots and analyzed using the i-Tree Eco model (2010)	(i.) The total amount of $PM_{2.5}$ removed annually by trees (ii.) Mortality reductions by trees were typically around (iii.) The net removal amounts per square meter of canopy cover (iv.) The average health benefits value per hectare of tree cover (v.) The value per tonne of $PM_{2.5}$ averaged	(i.) Varied from 4.7 tonnes in Syracuse to 64.5 tonnes in Atlanta, with annual values varying from $1.1 million in syracuse to $60.1 million in New York City (ii.) 1 person/yr per city, but were as high as 7.6 people/yr in New York City (iii.) Varied from 0.13 g m^2/ yr in Los Angeles to 0.36 g m^2/ yr in Atlanta (iv.) About $1,600, but varied from $500 in Atlanta and Minneapolis to $3,800 in New York (v.) $682,000, but varied from $142,000 in Atlanta to $1,610,000 in New York	(i) $1.21 million $66 million (iv) $1,761.07 ($550.33 – $4,182.53) (v) $751 thousand ($156 – $1,772 thousand)

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Economic measures	Adjusted WTP value (value in $US2016)*
				(vi.) The health benefits value per reduction of 1 μg/m^3	(vi.) Varied from $122 million in Syracuse to $6.2 billion in New York, with an overall average of $1.6 billion	(vi) 134 million – 6.82 billion (1.76 billion)
Nowak et al. (2014)	U.S. EPA's BenMAP program	United States	Computer simulations with local environmental data (2010)	Trees and forests in the conterminous United States Human health effects valued due to pollution reduction	Removed 17.4 million tonnes (t) of air pollution US$6.8 billion	$7.48 billion
Tallis et al. (2011)	Urban Forest Effects Model (UFORE)	The Greater London Authority (GLA), UK	Tree survey data, annual maps of PM10 distribution and observed/predicted meteorological conditions (2006)	Annual PM_{10} removal by urban canopy of GLA	852 - 2121 tonnes (0.7% and 1.4% of PM_{10} from the Urban boundary layer)	
Yang et al. (2008)	Dry deposition model	Chicago US	Chicago's Department of Environment for a list of green roofs resulting in a list of 170 green roofs. (2006)	Total air pollutants removed by 19.8 ha of green roofs in one year The annual removal air pollutants per hectare of green roof	1,675 kg (O_3 accounting for 52% of the total and NO_2 (27%), 85 kg	
Yang et al. (2005)	Urban Forest Effect model	Beijing, China	A field survey was conducted in June (2002).	Air pollution removal by trees in the central part of Beijing The carbon dioxide (CO_2) stored in biomass form by the urban forest	1261.4 tons of pollutants in 2002 About 0.2 million tons	
Energy savings						
Donovan and Butry (2009)	Regression analysis	Sacramento, California	460 single-family homes	The current level of tree cover on the west and south sides of houses in the `sample reduced summertime electricity use by 185 kWh (5.2%)	A london plane tree, planted on the west side of a house, can reduce carbon emissions from summertime electricity use by an average of 31% over 100 years	
McPherson and Simpson (2003)	Estimates of previous studies and computer simulation	California	Data from aerial photography were previously collected for 21 California cities	(i.) Existing trees are projected to reduce annual air conditioning energy use (ii.) Peak load reduction by existing trees	(i.) By 2.5% with a wholesale value of $485.8 million (ii.) Saves utilities 10% valued at approximately $778.5 million annually, or $4.39/tree	$677 million $1085 million ($6.12/tree)

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Economic measures	Adjusted WTP value (value in $US2016)*
				(iii.) The present wholesale value of annual cooling reductions for the 15-year period.	(iii.) $3.6 billion ($71/tree planted)	$5.01 billion ($98.96/tree)
Pandit and Laband (2010a)	a statistical model	Auburn, Alabama	160 residences. monthly electricity usage data (2007-2008)	Having dense shade at the sample mean (an average during the day of 19.30% of the residential structure)	Would save a home owner $21.22/month (9.3%) in electricity costs during the summer months, as compared to a home owner with no shade falling on the residence.	$23.65
*Mental and physical health benefits**						
Alcock et al. (2014)	Regression analysis with panel data	Estimation samples were limited to English residents	British Household Panel Survey with mental health data (1991 to 2008)	Individuals who moved to greener areas (n = 594) had significantly better mental health in all three post move years.	Individuals who moved to less green areas (n = 470) showed significantly worse mental health in the year preceding the move (P = 0.031) Moving to greener urban areas was associated with sustained mental health improvements	
Beyer et al. (2014)	Multivariate survey regression analyses	Wisconsin	Survey of the Health of Wisconsin (SHOW) database. (2008–2009, 2010 and 2011) (2,479 individuals nested in 229 Wisconsin Census Block Groups)	Higher levels of neighbourhood green space were associated with significantly lower levels of symptomology for depression		
Coombes et al. (2010)	Logistic regression	City of Bristol, England	Data from the 2005 Bristol Quality of Life in your Neighbourhood survey (6821 adults were combined with a comprehensive GIS database on green space)	The reviewed frequency of green space use declined with increasing distance	Respondents living closest to the type of green space classified as a formal park were more likely to achieve the physical activity recommendation and less likely to be overweight or obese	
Donovan et al. (2011)	Binary logistic regression	Portland, Oregon	5696 residents' (Birth certificates and tax records)	10% increase in tree-canopy cover within 50 m of a house	Reduced the number of small for gestational age births by 1.42 per 1000 births (95% CI—0.11–2.72)	

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Economic measures	Adjusted WTP value (value in $US2016)*
Francis et al. (2012)	Logistic regression analysis	Perth, Western Australia	From a cross-sectional survey - Perth residents in 2003 and December 2005	Residents of neighbourhoods with high quality public open space	Had higher odds of low psychosocial distress than residents of neighbourhoods with low quality public open space Public open space quality within a neighbourhood appears to be More important than public open space quantity	
Gidlöf-Gunnarsson and Öhrström (2007)	MANOVA analysis	Stockholm and Goteborg, Sweden	500 residents in urban setting	Better availability to nearby green areas	Reduced long-term noise annoyances and prevalence of stress-related psychosocial symptoms	
Sugiyama et al. (2008)	Stepwise logistic regression analyses	Adelaide, Australia:	Data from a mailed survey of adults (n = 1895) during 2003–2004	Those who perceived their neighbourhood as highly green	Had 1.37 and 1.60 times higher odds of better physical and mental health, respectively, compared with those who perceived the lowest greenness	
Zhang et al. (2015c)	Structural Equation Modelling	A medium-sized Dutch city in the Netherlands.	Mailed surveys in two neighbourhoods (n = 223)	Greater attachment to local green space and better self-reviewed mental health in the neighbourhood	Green space attachment is linked to mental health	

* US$1 = AU$1.355, 2016, Reserve Bank Australia

Ecological and environmental value of water

Rapid urbanization not only increases pressure on green infrastructure but also on local water bodies and ecosystems. For example, in Blacktown, Sydney, it has been estimated that for each million m^2 paved area 0.5 GL of extra water flows to the local creek system causing substantial problems for the local ecology and biodiversity (Liebman et al., 2015). The ecological and environmental value of water can be summarized in terms of water quality value, habitat conservation value, and aesthetics, and relevant studies are summarised below.

Water quality value

There are numerous studies on non-market values of water quality, and the most commonly used methods are stated preference methods and the travel cost method

Peng and Oleson (2017) reports the results of a choice experiment to understand beach recreationalists' preferences and willingness to pay for water quality and associated attributes at Hawaiian beaches. The study found that people were willing to pay US$35.71 extra per day at the beach to increase the visibility of water from 15 ft to 30 ft; and an additional $14.80 to increase visibility from 30 ft to 60 ft. The study also found a high preference for biodiversity; with people WTP $15.33 to improve coral reef cover from 10% to 25% and $4.89 to improve cover to 45%. The mean WTP for increasing the number of fish species from 9 to 18, was $7.14. People were also WTP $11.43 to reduce the number of days with bacteria exceedance above safe levels from 11 to 5 per year; and another $30.72 to reduce the number of bacteria exceedance above safe levels to zero.

In MacDonald et al. (2015) a choice experiment survey was used to estimate the total value of a project which could achieve multiple outcomes including: ensuring 25 days per year of water clarity, increasing seagrass area from 60% to 70% of the original area and protecting five reef areas. The study found that the total value of the project to households in the Adelaide, South Australia was AU$67.1 M.

In Viana et al. (2017) the travel cost method was used to estimate the average consumer surplus of the Channel Islands National Marine Sanctuary located in California, USA to private recreational boaters. The study found the value to be $48.62 per trip. The value was also found to be higher in locations with lower exposure to the prevailing winds, and greater species richness and abundance.

Alvarez et al. (2016) conducted a meta-regression analysis of water quality improvement in the United States. The study considers 19 studies (from 39 related studies) where Contingent Valuation, Travel Cost, and Choice Experiment methods were used. The study found that the predicted WTP is sensitive to the level of urbanization and population density. People living in urban areas are willing to pay more, however, as population density increased on average people are willing to pay less (i.e., residents in small urban areas are willing to pay the most). Projecting across the 67 Florida, US counties they found that the WTP for water quality improvement ranged from 4 cents to US$837 per person per year, for an improvement in water quality from level 5 (fishable) to level 7 (swimmable).

Van Houtven et al. (2007) a second meta-regression analysis of water quality. The study is based on 131 WTP estimates from 18 studies and found that for every 10% increase in the water quality index the WTP estimate would increase by 8%. Further, if the water quality improvement description included a recreational use description the mean WTP was higher by an average of $14 per person.

Klemick et al. (2016) conducted a meta-regression study that uses hedonic study estimates only. The study focus is the impact of total maximum daily pollution load on property prices in the Chesapeake Bay and the study found that at an aggregate level the near-waterfront property values could increase by roughly $400–$700 million in response to water clarity improvements.

Parsons et al. (2003) measured the economic benefits associated with recreation from improved water quality using choice experiments for six north-eastern states of the USA. In the study separate choice experiment models were used for fishing, boating, swimming, and viewing. For significant improvements in water quality, all four recreational activities were associated with benefits. Swimming and viewing were the activities that showed the highest gains, 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 (2011) 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 a choice experiment. They found that the mean annual WTP per household for water clarity from the current clarity (around 1 m) to: see up to 1.5, 2.0, and 4.0 m underwater were, respectively $4.17, $21.03, and $65.82.

Ge et al. (2013) is a meta-regression analysis on water quality improvement based on 38 US studies. The study found that for a 10-point (out of 100 points) additional change in the water quality index the mean WTP increases by $45. WTP was higher for lakes and estuaries than for rivers and higher for avoiding degradation than for making improvement. The study also found variation in WTP estimates across estimation methods. Studies using hedonic analysis had the highest mean WTP, followed by the travel cost method, and then the contingent valuation method. The initial condition of water quality and site size interact with each other, and a summary of the results are presented in Table 3.

Table 3: Predicted WTP for water quality improvement

Site type	WQI change			
	40 to 45	40 to 50	70 to 75	70 to 80
Small site (1 sq mile) (Little Spirit, IA)	115.14 (143.84)	137.52 (141.78)	35.12 (142.33)	57.50 (139.9)
Medium site (100 sq mile) (Lake Winnibigoshish, MN)	121.46 (141.32)	143.85 (139.29)	41.44 (139.62)	63.83 (137.23)
Big site (10,000 sq mile) (Great Lakes)	753.89 (210.71)	776.27 (213.64)	673.87 (197.95)	696.25 (200.84)

Source: Ge et al. (2013)

Note: WQI: Water Quality Index; Standard errors in parentheses, in 2010 US dollars, sample region: 50,000 sq miles

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 study 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 m) for the entire month of August.

Sale et al. (2009) assessed the value 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% 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, and Fleming and Cook (2008) estimate the recreational value of Lake McKenzie, Australia, using the travel cost method. Based on analysis of 1,360 survey responses the study 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.

A small number of studies have combined the contingent valuation method with the travel cost method to estimate recreational values. In Rolfe and Prayaga (2007) the value of recreational fishing at three major freshwater impoundments in Queensland, Australia, is estimated using both the travel cost method and the contingent valuation method. 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.

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, beyond the scope of the current review.

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.

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. (2014) 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 authors found that the mean WTP for visitors to prevent 10% and 25% 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.

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 children's 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% 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 (2008) 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, and 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 a 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.

Economic value of local stormwater management

There are some studies estimating non-market benefits of stormwater management. For example, Brent et al. (2016) conducted a choice experiment in Sydney and Melbourne to understand people's willingness to pay for different types of services: avoiding water restriction, improvement in local stream health, reduction in peak urban temperature and occurrence of flash flood reduction. The study found that people were willing to pay to: avoid water restrictions ($218 in Melbourne and $118 in Sydney); achieve improvements in local stream health ($278 in Melbourne and $104 in Sydney); and decrease peak urban temperatures ($81 in Melbourne and $47 in Sydney). However, people's willingness to pay to reduce the occurrence of flash floods was close to zero.

In Tapsuwan et al. (2014) people's willingness to pay for rainwater tanks and greywater systems in South East Queensland (SEQ) was estimated using the choice experiment method. The study found higher WTP for greywater systems ($1,700 – $14,100) compared to rainwater tanks ($800 - $7,400). The study also found that estimated values were lower than the installation and maintenance costs of these systems. However, in Zhang et al. (2015b), it was found that there is a significant positive effect of rainwater tanks on house prices in Perth, Australia. Specifically the study found that the presence of a rainwater tank on a property would add $6,700 to $18,000 to the median price of a typical house in Perth. This benefit is large enough to cover the total cost of installing and maintaining a tank.

Polyakov et al. (2016) assessed the changes in the amenity benefits of an urban drainage restoration project in Perth, Western Australia. After controlling for relevant factors it was found that the median home within 200 m of the restoration site had increased in value by an additional $17,000 to $26,000 after eight years. The study also found that the total benefit across all houses within 200 m of the project was more than enough to cover the cost of the restoration project.

In Mekala et al. (2015) the potential benefits of the rehabilitation of a 1.23 km stretch of upper Stony Creek in Melbourne are investigated. Based on secondary information, the study estimated the potential benefits of the project, including health benefits as around $75,000 per annum. The potential capitalized amenity benefit of the park was estimated at around $3.9 million.

It should be noted that it is not always possible to calculate net benefits of different stormwater management options due to lack of information on non-market values of the services provided by different options. In such cases, cost-effectiveness analysis can be used. An example set of cost estimates for different water sensitive urban design technologies is presented in Table 4.

Table 4: Water sensitive urban design life cycle costing data

Asset	Asset parameters	Construction[1]	Maintenance Establishment (First two years)	On-going	Renewal
Wetlands[2]	< 500 m^2 500 to 10,000 m^2 > 10,000 m^2	\$150/m^2 \$100/m^2 \$75/m^2		\$10/m^2/yr \$2/m^2/yr \$0.50/m^2/yr	No data
Sediment basin[2]	< 250 m^2 250 to 1,000 m^2 > 1,000 m^2	\$250/m^2 \$200/m^2 \$150/m^2		\$20/m^2/yr \$10/m^2/yr \$5/m^2/yr	Remove and dispose of Dry waste = \$ 250/m^3 Liquid waste = \$ 1,300/m^3
On-street rain gardens[2]	< 50 m^2 50 to 250 m^2 > 250 m^2	\$2,000/m^2 \$1,000/m^2 \$500/m^2		\$30/m^2/yr \$15/m^2/yr \$10/m^2/yr	Minor reset = \$50 to \$100/m^2
Bioretention basin	< 100 m^2 100 to 500 m^2 > 500 m^2	\$1,000/m^2 \$350/m^2 \$250/m^2		\$5/m^2/yr	
Tree pits[3]	< 10 m^2 10 to 50 m^2 > 50 m^2	\$8,000/m^2 \$5,000/m^2 \$1,000/m^2	Two to five times of on-going maintenance cost	No access issues = \$150 / asset / yr Traffic issues or specialist equipment required = \$ 500/asset /yr	
Grass swales and buffer strips[4]	Seeded – no subsoil drain Seeded – subsoil drain Turfed – no subsoil drain Native grasses established	\$15/m^2 \$25/m^2 \$20/m^2 \$35/m^2		\$3/m^2/yr	
Vegetated swales and bioretention swales[4]		\$150/m^2		\$5/m^2/yr	
In-ground GPTs	< 300 L/S 300 – 2000 L/S > 2000 L/S	\$50,000/asset \$150,000/asset \$250,000/asset		Inspection = \$100/visit Cleanout = \$1,000/visit	

Source: Melbourne Water (2013)

Note: [1]includes planning and design; [2]Area at normal water level; [3]Area of filter media at bottom of extended detention and [4]Total vegetated area. The cost estimates should be considered as a starting point only and represent best available information in 2013.

Climate change mitigation

It has been suggested that well-designed and developed green (and blue) spaces in landscapes have the potential to minimize climate change impact (Žuvela-Aloise et al., 2016, Demuzere et al., 2014, Gunawardena et al., 2017). Many of the non-market valuations studies reported under the green infrastructure section are also relevant to climate change mitigation options. However, most of the studies focusing on climate change impact have reported economic estimates that do not rely on the main environmental evaluation methods. As such, the following sub-sections provide a summary of existing information on economic costs and values of climate change mitigation options in terms of urban heat island mitigations, carbon sequestration and reduction of carbon emissions. Table 5 provides the list of studies with economic and non-economic values related to climate change mitigation benefits.

Urban heat island effect mitigation

The urban heat island effect is a major problem in many urban cities (Coutts et al., 2013). A heat island is a metropolitan area that experiences extreme temperatures especially during summer periods (Kim et al., 2016). This effect is caused by reflections from urban structures that absorb heat from the sun during daytime. Extreme heat events are events that could lead to high rates of mortality and morbidity in cities (Roldán et al., 2015), increased energy consumption and productivity losses

A study carried out in Singapore concluded that the cooling impacts of the parks are reflected through not only the lower temperatures in the parks but also the lower temperatures in the nearby built environment (Yu and Hien, 2006). The study was conducted in two big city green areas. One is the city's natural reserve—Bukit Batok Nature Park (BBNP) (36 ha) and the other is a neighbourhood park—Clementi Woods Park (CWP) (12 ha). The measurements were conducted at both vegetated areas and their surroundings. Another study in Singapore explored the impacts of green areas at the macro-level in mitigting heat island. The findings indicated a strong correlation between a decrease in temperature and the appearance of large green areas in the city (Wong and Yu, 2005). The results for Singapore are consistent with the findings of Susca et al. (2011) which looks at the impact of urban vegetation in heat island mitigation in four areas of New York City. Specifically the study found that there was an average of a 2ºC difference in temperatures between the most and the least vegetated areas.

Cooling is not only a function of vegetation and surface materials, but also dependent on the form and spatial arrangement of urban features. Studies have therfore also examined the impact of urban form and design on temperatures, for example (Middel et al., 2014). Studies have also looked at the possibility of using watered landscapes to manipulate urban heat island effects and estimate how much water it will take to cool local environments (Gober et al., 2009).

Some studies, such as Salata et al. (2017), have evaluated different configurations of green space -- cool roofs, urban vegetation and cool pavement -- in mitigting extreme heat events. This research suggests that changes to the configueration of green space can have a material impact on the performance of the area in terms of confort, and mitigating heat related health risks.

Žuvela-Aloise et al. (2016) used real case simulations to explore the best combination of heat mitigation strategies in Vienna, Austria. The results suggested that heat load mitigation measures have different efficiency depending on their location, with the main determining factors found to be the prevailing meteorological conditions and land use characteristics in the neighbouring environment.

Nakayama and Hashimoto (2011) examined the ability of water resources to reduce the urban heat island in Tokyo, Japan. The study focus was the relationship between the groundwater infiltration and the heat island effect and the result suggests that effective management of water resources has the ability to mitigate extreme heat conditions.

Guhathakurta and Gober (2007) study the effects of Phoenix's urban heat island on water use by single-family residences. The findings show that increasing daily low temperatures by 1°F is associated with an average monthly increase in water use of 290 gal for a typical single-family unit. These results suggest that planners should consider the effects on water demand as well as other environmental consequences when they evaluate growth strategies, and use incentives to encourage efficiency and sustainability

Carbon sequestration

Studies on carbon sequestration by urban green space show the importance of green infrastructure such as urban forests to mitigate climate change. For example, the value carbon sequestration by urban forests (about 400,000 trees) in Canberra during the period 2008–2012 was estimated at US$ 300,000 (Brack, 2002). Davies et al. (2011), also estimated carbon storage of a typical British city, Leicester, by surveying vegetation across the entire urban area. They found that urban vegetation stored 231,521 tonnes of carbon (16 kg C m^{-2} of urban area).

With the use of CO_2 reduction measures from subtropical Miami-Dade and Gainesville, USA, Escobedo et al. (2010) modelled carbon sequestration by trees to analyse policies that use urban forests to offset carbon emissions. The emission reduction due to carbon sequestration was reported as 3.6 tonnes/ha/yr in Miami-Dade and 5.8 tonnes/ha/yr in Gainesville.

Carbon storage and sequestration by urban forests in Shenyang, China was examined by Liu and Li (2012) in 2006. The C sequestration rate of the heavily industrialized city was estimated as 29,000 t/yr (RMB7.88 million, or $1.19 million). According to their estimates, carbon sequestration could offset 0.26% of the annual carbon emissions in Shenyang.

Nowak and Crane (2002), analysed field data from 10 USA cities and national urban tree cover data from 1996 and 1999 using Urban Forest Effects (UFORE) model. Urban trees in the coterminous USA, stored 700 million tonnes of carbon ($14,300 million value) with a gross carbon sequestration rate of 22.8 million t C/yr ($460 million/year). The national average density of carbon storage in the urban forest was 25.1 t C/ha.

The carbon storage by urban trees in Leipzig, Germany was estimated to be 316,000 Mg C at 11 Mg C/ha (Strohbach and Haase, 2012). The authors also noted that carbon storage in the city of Leipzig was in the lower range compared to cities in Europe, Asia and the USA.

Tran et al. (2017), investigated the WTP of Atlanta households to increase urban forests to mitigate climate change. The method used was the contingent valuation method, the survey was conducted in 2013, and the analysis found that households were willing to pay, in aggregate $1.05 million to $1.22 million per year to increase the amount of urban forests. Kim et al. (2016), investigated residents’ WTP on the heat island-mitigating functions of urban forest in Korea via a choice experiment and found respondents were willing to pay $56.68–76.59 for an increase of the urban forest by 1 m^2.

Reduced carbon emissions

According to Akbari (2002), a tree planted in Los Angeles would avoid the combustion of 18 kg of carbon, annually. It was then estimated that trees can potentially save about $270 M per year in Los Angeles, and can reduce peak power demand by 0.9 GW. Of the $270 M annual savings, about $58 M represent direct energy savings, $35 M indirect energy savings, and $180 M savings because of the reduction in smog concentration.

A case study of the value of the Canberra urban forest with particular reference to pollution mitigation was estimated at US$20–$67 million (or $66–$223/resident) between 2008 and 2012 (Brack, 2002).

Escobedo et al. (2010), estimated the effects of urban forests on building energy use due to shading and climate regulation. Avoided carbon emissions due to energy savings as a result of shade were estimated to be 0.65 tonnes/ha/year in Gainesville and 0.166 tonnes/ha/year in Miami-Dade. Avoided carbon due to climate regulation was 0.70 tonnes/ha/year for Gainesville and 0.173 tonnes per ha/year for Miami-Dade.

The value of services provided by trees in Allan Gardens, a historic public park in downtown Toronto, Canada was examined by Millward and Sabir (2011). On a per-tree basis, CO_2 removal benefits derived from a Scotch Elm was estimated to be $10 per tree. Silver Maple and Black Walnut reduced carbon emissions worth $6 per tree, while Norway Maple reduced emissions worth $5 per tree.

Table 5: Climate change mitigation

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Non-economic measures of benefits	Economic measures of benefits	Adjusted WTP value (value in $US2016)*
Akbari (2002)	Estimate from previous studies	Los Angeles Baton Rouge, Sacramento, and Salt Lake City,		A tree planted in Los Angeles Planting an average of four shade trees per house (each with a top view cross section of 50 m^2)	Avoids the combustion of 18 kg of carbon annually, even though it sequesters only 4.5–11 kg Would lead to an annual reduction in carbon emissions from power plants of 16,000, 41,000, and 9000 t, respectively (the per-tree reduction in carbon emissions is about 10–11 kg per year) Urban tree planting can account for a 25% reduction in net cooling and heating energy usage in urban landscapes		
Brack (2002)	The statistical models	Canberra urban forest	400.000 trees in Canberra (2008-2012)			The planted trees are estimated to have a combined energy reduction, pollution mitigation and carbon sequestration value of US$20–67 million during the period 2008–2012	$21 – $70 million
Davies et al. (2011)	Statistical tests	Leicester	Vegetation survey	Total Carbon storage	231,521 tonnes (95% ci = 195,914–267,130) of carbon is stored within the above-ground vegetation across the city (equating to a mean figure of 3.16 kg C m^{-2} of urban area)		
Derkzen et al. (2017)	Multidimensional CV	Rotterdam, the Netherlands	(in 2014)	About two thirds of respondents were willing to pay for green infrastructure measures as a tax.		WTP $15 per household per year as green tax	$15.21

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Non-economic measures of benefits	Economic measures of benefits	Adjusted WTP value (value in $US2016)*
de Koning et al. (2017)	Agent based modelling	Greenville,NC,, US	Property market data (9793 records between 1992 and 2002) and income and housing budget data	The bias in marginal implicit price of flood risk ranges between 4.2% and 9.7%.		Clear differences in the marginal implicit price of flood risk among different behavioural risk perception models	
Escobedo et al. (2010)	Urban Forest Effects (UFORE) Model	Subtropical forests Miami-Dade and Gainesville, USA	Field data (2005-2008)	Emission reduction in Miami-Dade- 3.6 (tonnes/ha/yr) Gainesville 5.8 (tonnes/ha/yr)	Urban tree sequestration offsets CO_2 emissions and, relative to total city-wide emissions, is moderately effective at 3.4% and 1.8% in Gainesville and Miami-Dade, respectively		
Hungate et al. (2017)	Experiments	North American grassland	Plant, soil, and ecosystem carbon storage data from two grassland biodiversity experiments		Increasing species richness from 1 to 10 had twice the economic value of increasing species richness from 1 to 2. The marginal value of each additional species declined as species accumulated, reflecting the nonlinear relationship between species richness and plant biomass production		
Kim et al. (2016)	CE	Summer Season in Korea	448 people from metropolitan regions of Seoul, Busan, Incheon, Kwangju, Daejeon, Ulsan, and Daegu Sept. 2010	Marginal willingness to pay for every increase of the urban forest by 1 m^2		$56.68–76.59	$62.39 – $84.20
Kim et al. (2017)	HD	Woomyeon Nature Park (WNP) in Seoul, Korea which experienced a catastrophic landslide disaster in 2011	sales data of the Ministry of Land, Infrastructure, and Transport (MLIT), Korea from 2008 to 2014	Housing market premiums		Have fallen by up to 11.3% since the event due to the risk of landslide	

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Non-economic measures of benefits	Economic measures of benefits	Adjusted WTP value (value in $US2016)*
Lafortezza et al. (2009)	ANOVA	green spaces in Italy and the UK	800 respondents	Longer and frequent visits of green spaces	Generate significant improvements of the perceived benefits and wellbeing among users during the periods of heat stress		
Liu and Li (2012)	Biomass equations, images	Shenyang North-Eastern China	Field survey data and urban forests data derived from high-resolution Quick Bird (2006)	Urban forests in areas within the third-ring road of Shenyang stored C sequestration rate		337,000 t C (RMB92.02 million, or $13.88 million) 29,000 t/yr (RMB7.88 million, or $ 1.19 million). The C stored by urban forests equalled to 3.02% of the annual c emissions from fossil fuel combustion and C sequestration could offset 0.26% of the annual C emissions in Shenyang	$16.52 million $1.41 million
Nowak et al. (2017)	Five types of analyses	United States	Field data on urban trees, urban/commu-nity tree and land cover maps (2006-2010)	(i.) Trees and forests in urban/community areas in the conterminous United States (ii.) Reduce energy use for heating (iii.) Avoid thousands of tonnes of emissions of several pollutants (iv.) Average reduction in national residential energy use due to trees (v.) The greatest avoided emissions nationally due to energy conservation came from (vi.) The greatest associated savings from (vii.) avoided emissions		(i.) Annually reduce electricity use by 38.8 million MWH ($4.7 billion) (ii.) By 246 million MMBTUS ($3.1 billion) (iii.) Valued at $3.9 billion per year (iv.) 7.2% (v.) CO_2 (43.8 million tonnes), followed by SO_2 (113,000 t) and NO_x (39,000 t) (vi.) CO_2 ($1.8 billion), followed by SO_2 ($1.0 billion) and pm2.5 ($638 million)	(i) $5.17 billion (ii) $3.41 billion (iii) 4.29 billion (vi) 1.98 billion - $1.10 billion – $702 million

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Non-economic measures of benefits	Economic measures of benefits	Adjusted WTP value (value in $US2016)*
				(viii.) The overall value for avoided emissions nationally was		(vii.) $3.9 billion per year	(vi) $4.29 billion
Nowak and Crane (2002)	Urban Forest Effects (UFORE) model	USA	Field data from 10 USA cities and national urban tree cover data 1996 and 1999	Urban trees in the coterminous USA	The national average urban forest carbon storage density is 25.1 t c/ha, compared with 53.5 t c/ha in forest stands.	Currently store 700 million tonnes of carbon ($14,300 million value) with a gross carbon sequestration rate of 22.8 million t c/yr ($460 million /year)	$20,600 million ($921 million)
Roldán et al. (2015)	Autoregressive integrated moving average model	Zaragoza, Spain	Mortality data (Public Health Directorate of the Govt. of Aragón; and, temperature data (Foundation for Climate Research and the State Meteorological Agency.) 2002–2006	Mortality showed a statistically significant increase when the daily maximum temperature exceeded 38°C. A Relative Risk was 1.28 with a 95% confidence interval (95% CI:1.08–1.57)		Heat-attributable deaths were estimated for the period 2002–2006, and the in-hospital estimated cost of these deaths reach € 426,087 (95% CI €167,249–€688,907)	$652,006 ($255,927 - $1,054,179)
Susca et al. (2011)		New York City;	Survey: 2008 - 2009	Monitoring the urban heat island	Found an average of 2°C,difference of temperatures between the most and the least vegetated areas, ascribable to the substitution of vegetation with man-made building materials		
Soares et al. (2011)	The computer tool i-Tree STRATUM	Lisbon, Portugal	An inventory of all 33,232 trees was completed in 2003 under supervision of the Gardens Department of the Municipality of Lisbon	(i.) For every $1 invested in tree management, the value of (ii.) Energy savings (iii.) CO_2 reduction (iv.) Air pollutant deposition (v.) Stormwater runoff reduction (vi.) Increased real estate value		(i.) $4.48 in benefits (ii.) $6.20/tree (iii.) $0.33/tree (iv.) $5.40/tree (v.) $47.80/tree (vi.) $144.70/tree	(i.) $5.84 (ii.) $8.09 (iii.) $0.43 (iv.) $7.04 (v.) $62.35 (vi.) $188.74

Author	Method	Location	No. of records (respondents)	Definition of marginal change	Non-economic measures of benefits	Economic measures of benefits	Adjusted WTP value (value in $US2016)*
Strohbach and Haase (2012)	Allometric equations, using a bootstrap method	Leipzig, Germany	Stratified random sampling across 19 land cover classes (2009)	Canopy cover was approximately 19% of the city area Leipzig's above-ground carbon storage	316,000 mg C (at 11 mg C/ha)		
Tran et al. (2017)	Contingent valuation method (CVM)	Atlanta, Georgia, USA	Mail-based survey was developed and administered using Dillman's Tailored Design Method (2013)	Households WTP for urban forests as a climate change mitigation method.		$1.05 - $1.22 million per year, or $5.24 - $6.11 million over a five-year period	$1.08 – $1.25 million ($5.40 – $6.29 million)
Wong and Yu (2005)		Green areas at macro-level in Singapore	Island –wide temperature maps developed form the data derived from a mobile survey	A strong correlation between the decrease of temperature and the appearance of large green areas in the city.	The maximum difference of 4.01c was observed between well planted area and the CBD area		
Yu and Hien (2006)	Two simulation programmes using TAS and Envi-met	Bukit Batok Nature Park (BBNP) and another neighbourhood park in Singapore	Field measurements and localized weather data (11 January to 5 February 2003 and 16 June to 1 July 2003)	Maximally, 1.33°C difference of average temperature was observed at locations around the parks.	The temperature difference was caused by green areas and it may lead to savings of cooling energy and thermal comfort for residents. The cooling impacts of the parks are reflected through not only the lower temperatures in the parks but also the lower Temperatures in the nearby built environment		
Žuvela-Aloise et al. (2016)	Real case simulations	Vienna	Combined dataset of 32 different land use types and meteorological data	With the application of several heat load mitigation measures such as decrease in building density by 10% and pavement by 20%, enlargement in green and water spaces by 20%, it is possible to achieve substantial cooling effect with heat load reduction of −10 SU or more with a relatively small change in infrastructure	The modelling results show that equal heat load mitigation measures may have different efficiency dependent on location in the city due to the prevailing meteorological conditions and land use characteristics in the neighbouring environment		

* US$1 = AU$1.355, 2016, Reserve Bank Australia

Nowak et al. (2017) estimated the reduction in energy costs and avoided power plant emissions due to trees and forests in urban/community areas in the 48 contiguous states of the United States using data from 2006-2010. According to the study estimates, the annual reduction of electricity was 38.8 million MWh ($4.7 billion), with energy use for heating decreased by 246 million MMBtus ($3.1 billion). Trees helped in avoiding emissions of several pollutants valued at $3.9 billion per year including carbon emissions.

Soares et al. (2011), evaluated the benefits of urban trees in Lisbon, Portugal using the computer tool i-Tree STRATUM. Carbon emission reductions per tree were valued at $0.33/tree. The value of energy savings were reported as $6.20/tree. The stormwater runoff reduction impact was valued at $47.80/tree.

Non-point source pollution

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 non-point 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 non-point 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 non-point source pollution largely focused on estimating the damage costs caused by the pollution and/ or the environmental and public health risks created by pollution (Haynes and Georgianna, 1989). As it is hard to separate the influence of point source pollution from non-point 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 non-point source pollution in the USA could be as high as 2.7% of GDP.

In terms of understanding the non-point source pollution problem, Ventura and Kim (1993) suggest that it can be better 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.

Value of pollution removal based on abatement cost

There is some information on the value of removing pollutants from stormwater in Australia. Payne et al. (2015) reviewed that removal of one kilogram of nitrogen is valued at $6,645 (2014) based on past stormwater treatment works in Melbourne. On the other hand, for Sydney, the Department of Environment and Conservation (2006) provided information on removing pollutants using a hypothetical constructed wetland: Total Suspended Solids ($2.50/kg in 2012 dollars), Total Nitrogen ($625/kg) and Total Phosphorous ($2,501/kg). Liebman et al. (2015) reviewed treatment costs of removing major pollution using an off-site, precinct scale approach to managing stormwater as an alternative to the current on-site approach for each new development for Blacktown City Council, Sydney (Table 6).

Table 6: Cost of removing pollutants from stormwater ($/kg) using off-site treatment in Blacktown City Council, Sydney

Pollutant	Capital cost remove	Discounted maintenance cost (at 5% discount rate)	50 year whole of life cycle cost to remove
Total Suspended Solids (TSS)	62	20	82
Total Phosphorus (TP)	41,400	15,000	56,400
Total Nitrogen (TN)	5,900	2,400	8,300

Source: Liebman et al. (2015)

Hall (2012) provided a comprehensive analysis of different abatement options to remove urban water pollution in Brisbane where the cost-effectiveness of various options is calculated based on marginal cost estimates. Their estimates for cost-effectiveness of selected management options for removing two major pollutants (Total Nitrogen and Total Phosphorous) are presented in Table 7. As can be seen, the cost-effectiveness estimates vary with the size and type of technologies. The difference is due to economies of scale and effectiveness of removing pollutants. The cost gradually increases for smaller plants. Large wastewater treatment plants with biological nutrient removal could be very cost-effective compared to some other technologies. However, a comparison of low and high estimates for individual combinations of size and technology reveals that in many cases the range is quite large, which indicates a high level of uncertainty associated with some these estimates.

Table 7: Abatement cost per tonne of pollutant (AU$2010) for selected options

Technology	Plant size (MLd) / Project	TN		TP	
		L	H	L	H
Biological nutrient removal	0-0.379	12,524	5,566	166,699	6,262
	3.79 – 37.9	3,129	1,391	4,172	1,565
	>37.9	1,056	469	1,408	528
Reuse on eucalypt sawlog plantation using effluent from wastewater treatment plants	0.5	35,142		312,370	
	1	27,558		244,961	
	2	18,745		166,623	
	5	15,807		140,511	
	10	13,429		119,372	
	20	11,206		99,611	
	50	9,943		88,381	
	100	9,385		83,424	
Tertiary filtration	5	131,507	43,836	54,795	24,353
	10	119,178	39,726	49,658	22,070
	20	104,795	34,932	43,664	19,406
	50	69,863	23,288	29,110	12,938
	100	36,986	12,329	23,117	10,274
Stormwater harvesting	5 ML/Yr	12,810,000	1,490,000	67,300,000	6,730,000
	10 ML/Yr	5,640,000	660,000	29,600,000	2,960,000
	20 ML/Yr	2,750,000	320,000	14,400,000	1,440,000
	50 ML/Yr	730,000	90,000	3,900,000	390,000
WSUD – Bioretention	Greenfield residential (sloping topography)	106,130		429,834	
	Greenfield residential (greening topography)	255,442		1,058,258	
	Townhouses	1,497,703		5,706,057	
	Urban renewal	157,477		794,356	
	Commercial development	810,707		4,724,846	
	Industrial development	568,887		2,386,216	
5-kL Rainwater tank Yield scenario	70 kL/Yr	160,000		2,490,000	
	50 kL/Yr	320,000		5,130,000	
	30 kL/Yr	710,000		11,300,000	
WSUD – Swales		454,129		21,690	

Source: Hall (2012)

Note: Pollution abatement costs were calculated as the net present value of capital and operating costs divided by the pollution abated over the period of analysis; 3% discount rate for a 20-year period analysis. In estimating separate cost-effectiveness for removing individual pollutants two-thirds of the cost has been allocated to nitrogen removal and one-third to phosphorous removal.

Polyakov et al. (2017) provides a systematic cost-effectiveness analysis of the management of nutrient emission in the Swan-Canning catchment, and considers efficient abatement policy for an urban catchment. The study uses a comprehensive optimization model that mimics the decision of a single regulator who tries to minimize the cost of achieving pollution target by spreading actions across sub-catchments and time periods. The actions considered were: education of households, soil amendment, removal of septic tanks, investment in constructed wetlands, and banning standard fertilizers further to the restrictions introduced in 2010 on the phosphorus content of domestic fertilizers. The study compared the following scenarios: allow all abatement actions except banning standard fertilisers, both with and without the amenity value of constructed wetlands considered and allow banning standard fertilisers.

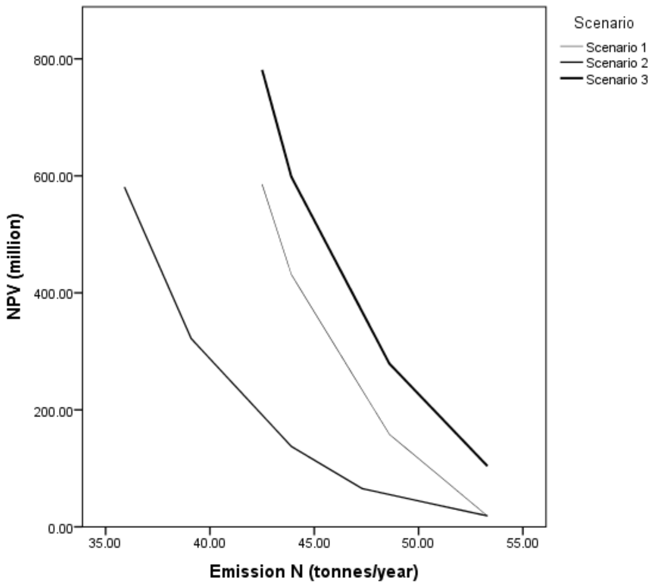


Figure 2: Abatement cost against nitrogen emissions (Polyakov et al., 2017)

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). Two relevant Australian studies are Kragt and Bennett (2009) and Bennett et al. (2008). Kragt and Bennett (2009) studied community preferences for natural resource management options in the George catchment in north-eastern Tasmania. They administered the survey in different sub-sample locations in Tasmania to assess the trade-offs that respondents were willing to make between environmental attributes and costs. They found that the respondents were, on average, willing to pay between AU$3.47 and $5.11 for a km increase in native riverside vegetation and between $7.10 and $12.42 per species for the protection of rare native plants and animals.

Bennett et al. (2008) studied the benefits associated with improvements in river health in Victoria using choice experiments. Monetary values were estimated for four attributes of environmental improvement: the percentage of pre-settlement fish species and populations; the percentage of the river's length with healthy vegetation on both banks; the number of native water bird and other fauna species with sustainable populations; and the percentage of the river suitable for primary contact recreation without threat to public health. Only the in-catchment respondents and Melbourne respondents were willing to pay for water quality improvements. The amount was approximately $2 for a 1% increase in the length of the river suitable for primary contact recreation.

The valuation of the environmental attributes of NSW rivers was considered in Bennett and Morrison (2001), where five rivers were studied: the Bega, Clarence, Georges, Gwydir and Murrumbidgee. The value estimated for an increase of 1% in the length of the river with healthy native vegetation and wetlands was in the order of $1-$2 (AU) per respondent. For an additional fish species, the value estimated was, on average, around $2-$3 and for water bird and other fauna species the average respondent was willing to pay approximately $1-$2. The value to improve the water quality up to the point where the river was swimmable throughout, would be an additional $35 (on average).

Flood hazard reduction

Flooding is a major natural hazard faced by many urban areas in Australia. From the catchment level perspective, there are two types of flood hazard: urban flooding and rural flooding. Both stormwater and stream 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.

According to an estimate based on a review of natural disasters in Australia occurring between 1967 to 2005, flood events are the most costly natural disaster (Middelmann ‐ Fernandes, 2010). In terms of the relative importance of stormwater and mainstream flow to flooding, SCARM (2000) reported that urban flooding caused by stormwater overflow, on average, represents 11% of the total flooding costs in Australia.

The costs from flooding can be grouped into five categories: direct costs, business interruption costs, indirect costs, intangible costs and risk mitigation costs (Meyer et al., 2014). Direct costs are related to the damage to property from direct physical impacts. Business interruption costs are related to the loss in productivity from the inability to carry out usual activities in the areas directly impacted by the flooding. Indirect costs happen outside of the direct impact area and over a long period of time, and are a flow-on consequence of direct costs and business interruption costs. Intangible costs are related to the non-market impacts, and these costs are not easily measurable (e.g., environmental or health costs). Risk mitigation costs are related to the preventive measures taken by people to reduce or minimize the flood impact. These costs could be direct, indirect or intangible. (Table 8).

Table 8: Cost categories of flood

Cost types		Tangible cost	Intangible (non-market) costs
Damage costs	Direct	Physical damage to assets: buildings, contents, infrastructure	Loss of life Health effects Loss of environmental goods
	Business interruption	Production interruptions because of destroyed machinery	Ecosystem services interrupted
	Indirect	Induced production losses of suppliers and customers of companies directly affected by the hazard	Inconvenience of post-flood recovery Increased vulnerability of survivors
Risk mitigation costs	Direct	Set-up infrastructure Operation and maintenance costs	Environmental damage: due to development of mitigation infrastructure or due to change in land use practices
	Indirect	Induced costs in other sectors	

Source: Meyer et al. (2014)

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 sets can be used to generate estimates of the potential flood damage risks in certain areas (Thompson et al., 1997). Lovelace and Strauser (1998) 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 the flood. FEMA (2012a) 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 would have been directed 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. Following Smith (1994) this approach 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, 2012b) 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 caused by floods under various conditions.

In Australia, some attempts have been made to develop stage-damage curves. For example, The State of Queensland (2002) provided stage-damage relationships for residential (Table 9) and commercial properties (Table 10). It can be observed that damage cost increases with higher flood depth. It should however be noted that the state-damage curve captures only the direct costs. Often a rule of thumb is used to calculate indirect costs. For residential properties, it is assumed that indirect damage is 15% of the direct damage whereas for commercial properties the indirect cost is estimated at 55% of the direct cost. However, indirect damages do not consider intangible costs and risk mitigation costs. Further, these functions only show potential damage as the actual damage could be lower or higher depending on the preparedness of the community.

Table 9: Stage-damage relationships for residential properties

Depth over floor level (m)	Small house: < 80 m^2 and/or 1–2 bedrooms	Medium house: 80–140 m^2 and/or 3 bedrooms	Large house: > 140 m^2 and/or 3+ bedrooms
0	905	2,557	5,873
0.1	1,881	5,115	11,743
0.6	7,370	13,979	25,351
1.5	17,379	18,585	32,276
1.8	17,643	18,868	32,768

Source: The State of Queensland (2002)
Note: The numbers are AU$ in 1992

Table 10: Stage-damage relationships for commercial properties in Queensland

Depth over floor level	Value class				
	1	2	3	4	5
Small commercial properties (<186m^2)					
0	0	0	0	0	0
0.25	2,202	4,405	8,809	17,618	35,237
0.75	5,506	11,011	22,023	44,046	88,092
1.25	8,258	16,518	33,034	66,069	132,137
1.75	9,176	18,352	36,705	73,410	146,819
2	9,726	19,454	38,907	77,814	155,628
Medium commercial properties (186 - 650m^2)					
0	0	0	0	0	0
0.25	6,975	13,948	27,896	55,791	111,583
0.75	16,884	33,768	67,537	135,074	270,147
1.25	25,693	51,387	102,773	205,574	411,094
1.75	28,445	56,893	113,785	227,570	455,140
2	30,281	60,564	121,126	242,252	484,504
Large commercial properties (>650m^2)					
0	0	0	0	0	0
0.25	7	15	32	61	122
0.75	39	78	154	308	619
1.25	81	162	326	649	1,297
1.75	132	267	533	1,065	2,129
2	159	318	636	1,272	2,545

Source: The State of Queensland (2002)

Note: Examples under individual value classes: 1: Florist, garden centres, sports pavilions, consulting rooms, vehicle sales areas, schools, churches; 2: Cafes / takeaway, service stations, pubs, second hand goods, clubs; 3: Chemists, musical instruments, printing, electronic goods, clothing; 4: Bottle shops, cameras, and; 5: Pharmaceuticals, electronics. The numbers are AU$ in 1992.

Evaluate flood risks and protection measures

There is some literature that estimates 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. (2009) 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.

Streiner and Loomis (1995) calculated the property value changes in the USA following urban stream restoration measures, including flood protection measures. The authors found that flood damage reductions and stream stabilizations together can add around 3 to 5% 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.

Daniel et al. (2009) provided a meta-analysis of economic impact from reduced flooding risk. They used 19 studies from the US in their analysis and found that an increase in the probability of flood risk by 1% in a year could result in -0.6% reduction in prices for an otherwise similar house. It was observed that with time the marginal willingness to pay for reduced risk exposure has increased and higher income areas have slightly lower willingness to pay. However, these estimates could be sensitive to the interactions of amenity benefits and risk exposure from living closer to water.

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 over-estimate 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 non-property 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.

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, 2005). 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.

Other methods

Various other methods are often used to calculate monetary values of impact of flood, e.g., cost-of-illness and value of lost-production. The cost-of-illness (COI) approach is used to calculate the total cost of diseases occurred due to a natural disaster. This approach includes several categories of direct and indirect costs: personal medical care costs for diagnosis, procedures, drugs and inpatient and outpatient care, non-medical costs, such as the costs of transportation for treatment and care, non-personal costs like those associated with information, education, communication and research, and finally income losses. The value of lost production is similar to the cost-of-illness approach which focus on the loss of income (Lekuthai and Vongvisessomjai, 2001).

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. Groundwater resources support complex ecosystems and agricultural production. In some cases, they have been integrated into the potable water supply for cities. Adjusted to 2016 dollars, it has been estimated that groundwater contributes around $36 billion per annum to the Australian economy (Chong and Sunding, 2006). If non-market values for ecosystem services were also included the contribution would be much higher.

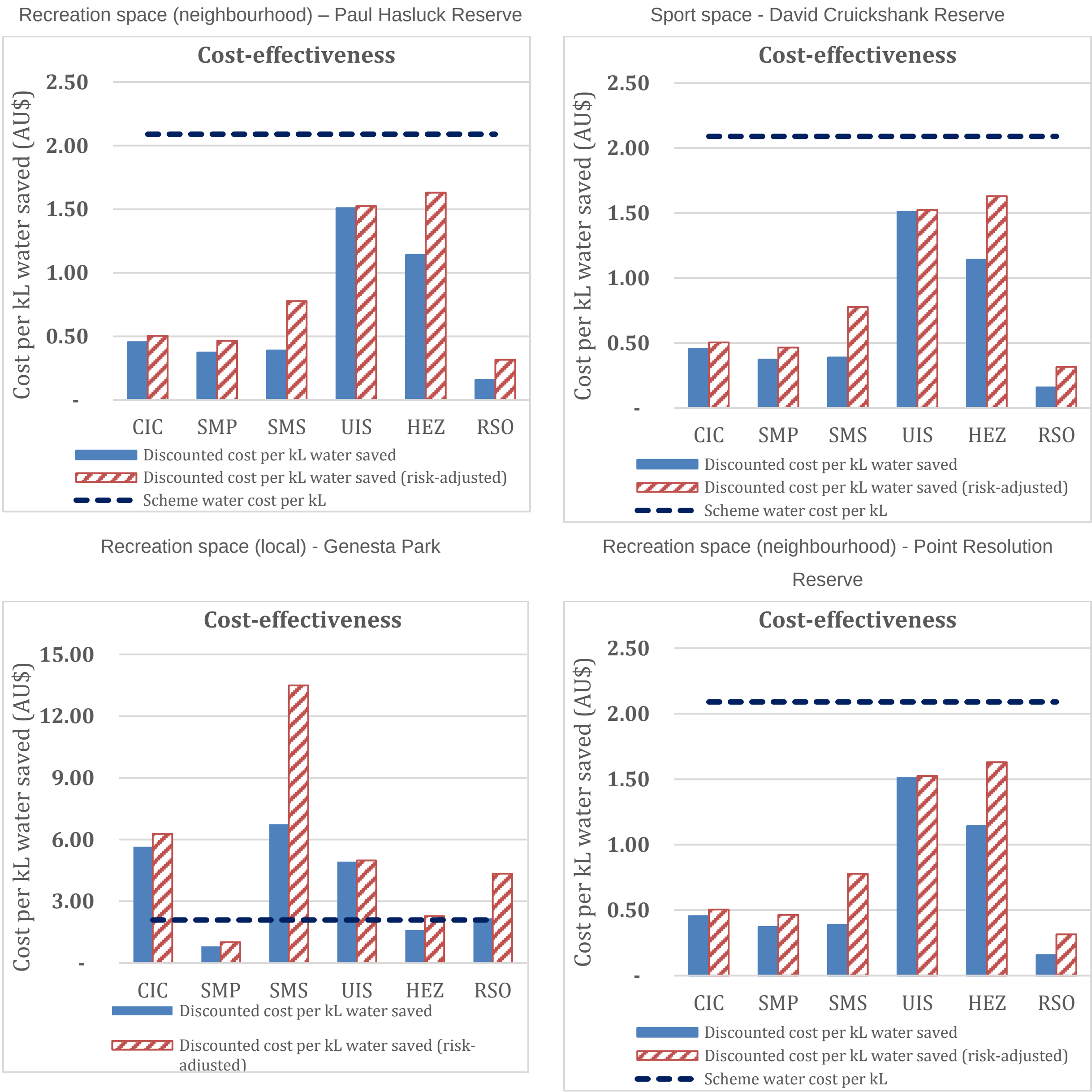
In Australia, and many other countries, however, groundwater is being extracted well beyond sustainable levels, placing them at risk (Harrington and Cook, 2014). For example, the groundwater storage in the Murray-Darling Basin, covering roughly 14% of Australia, lost around 100,000 GL between 2000 to 2007 due to climate change and over-extraction (Ranjan, 2014). As aquifers are out of sight, groundwater protection is a management area that has not always been a priority. However, there is some information on the value of groundwater use in Australia and elsewhere. Relevant information on the direct and indirect use values of groundwater is presented below.

Direct use values of groundwater

Economic valuation studies of the use value of groundwater focus on the role of groundwater as a water supply source. There have been a number of studies which have estimated the aggregated value of groundwater systems in Australia. The results show higher value for public water supply and use in industry applications. However, as these estimates only reflect the consumptive use value they are partial estimates. They do not capture the “non-extractive” or “option” values of groundwater. An example of non-extractive use would be use of water in forestry. An example of option value is when the availability of groundwater is considered during long-term planning even if the water is not used currently. For example, an irrigator with a surface water allocation may decide to plant long-lived horticulture products if they know that there is also a reserve groundwater supply. Based on a wide variety of data sources (Deloitte Access Economics Pty Ltd, 2013) estimated the value of some major groundwater direct uses as:

- Agriculture: irrigation: $30/ML - $500/ML
- Mining: $500/ML - $5,000/ML
- Urban water supply: $1,000/ML - $3,000/ML
- Households: $1,400/ML - $6,400/ML
- Manufacturing and other industries: $1,000/ML – $3,000/ML

We also reviewed results from two other studies on the value of groundwater in Perth, WA. Mennen (2017) analysed various water use efficiency improvement strategies to manage public open space. Based on empirical data, she conducted cost-effectiveness analysis of six water savings techniques for four parks of different sizes in Perth (Figure 3). She found that even though there is substantial variation in cost-effectiveness between different techniques it is possible to maintain the same level of Public Open Space (POS) quality even in the face of water supply reduction by adopting more efficient techniques. For larger parks, it is cheaper to improve efficiency than securing water from scheme water and the most cost-effective technique is the use of ‘Rain shut off devices’. In contrast, for a small local park ‘Improving soil moisture properties’ and ‘Soil moisture sensors’. Again, except for one or two techniques it is cheaper to improve efficiency.



Note: CIC: Central irrigation control with weather station; SMP: Soil moisture properties; SMS: Soil moisture sensors; UIS: Upgrade system 85% DU; HEZ: Hydrozoning/ ecozoning and RSO: Rain shut off

Figure 3: Cost-effectiveness of six techniques for four parks from Mennen (2017)

In the second Western Australian study, Iftekhar and Fogarty (2017) estimated the loss in gross and net revenue for horticulturists from reduction in groundwater extraction rights or allocation in Gnangara, WA. Based on their simulation analysis, they found that the average per hectare total return and net cash return were $27,248 and $7,104, respectively and with a 25% reduction in water allocations the net allocation will fall by $1,000 per ha, or 13.1%. However, there is large difference in loss between different sizes of farms. For example, the expected loss in net revenue per ha for a 10 ha farm was around three times the expected loss per ha for a 1 ha farm; and the expected loss per ha for a 25 ha farm was around five times the expected loss per ha for a 1 ha farm.

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 (see Table 11) for relevant literature). 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.

Hérivaux and Rinaudo (2016), investigated two aquifers: Meuse alluvial aquifer (MAA) in Belgium and Lower Triassic Sandstone (LTS) in France. They conducted two contingent valuation surveys of 530 respondents from the city of Liege in Belgium and 650 respondents from the Lorraine region in France. The study estimated the WTP of people to protect groundwater which included both use and non-use values. On average households were willing to pay €40 per year over 10 years to protect each aquifer.

Vo and Huynh (2017) studies the willingness to pay to protect groundwater over a five year period in the Mekong Delta, Vietnam. The contingent valuation method was used and the study sample size was 598. The study found that the willingness to pay to protect the groundwater over next five years was US$6.74 per year.

Some studies have investigated how much people would be willing to pay to improve the water quality of water bodies by restoring the groundwater. For example the contingent valuation study of Rinaudo and Aulong (2014) estimated the WTP of residents to restore groundwater in the upper Rhine valley alluvial aquifer between Germany and France in 2006. The WTP for improving water quality to the level of potable water was estimated as €42/year/household over 10 years. The WTP to improve the water quality to pristine water was €76/year/household over 10 years.

Using the choice experiment method Kountouris et al. (2014) estimate the WTP to improve water quality by protecting groundwater from nitrate pollution in Czestochowa Region of Poland in 2011. In the study respondents were found to be willing to pay €13.09–13.20 in addition to their current water bill to secure better water quality.

Tentes and Damigos (2012) examined households WTP for the remediation of polluted groundwater in Asopos river basin aquifer in Greece using CV technique. The WTP estimates ranged from €180- 239 per household per year.

Using data from a contingent valuation study conducted in Albacete, Spain, Rupérez-Moreno et al. (2015) examined people's preferences to improve ecological status of the water bodies of an aquifer. The study found that users were willing to pay € 35.88 per year while the value for non-users was €14.86 per year.

Table 11: Groundwater valuation surveys

Author	Method	Location	No. of surveys	Study	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2016)*
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	$10.45 million-$52.27 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	$2.08-$18.08
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	$256.95
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	$447.52 - $722.03
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	$1,115.81
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	$101.39
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	$53-$516.09
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	$37.84-$48.71
White et al. (2001)	CV	Waimea Plains, Nelson, New Zealand	180	Estimate the value of the groundwater resource in terms of benefits for irrigation,	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%	$106.1-$174.05

Author	Method	Location	No. of surveys	Study	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2016)*
				commercial/industrial use and bulk water supply	reduction in groundwater extraction is $183 per household per year	
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	$305.66-$1383.94
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	$107.67-$387.41
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	$60.55-$109.45
Martínez-Paz and Perni (2011)	Production function method and CV	Gavilan Aquifer, Spain	309	Estimate the total economic value of groundwater resources	0.381 €/m^3 as the value of groundwater for agriculture; 0.010 €/m^3 as the value of groundwater for recreational activities; 0.063 €/m^3 as the value of groundwater for environmental functions	0.01$/$m^3-0.52/m^3
Tentes and Damigos (2012)	CV	Asopos river basin aquifer in Greece	310 households by personal interviews (2009)	Households WTP for the remediation of polluted groundwater in the area of interest ranges	180€ - 239€ per household per year.	$283.62-336.62
Marella and Raga (2014)*	CV	Northern Italy	150 residents	Economic values of the individual willingness to pay (WTP) for landfill mining and the subsequent creation of a public park	Mean WTP was €196	

Author	Method	Location	No. of surveys	Study	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2016)*
Kountouris et al. (2014)	CE	Czestochowa Region of Poland	400 (2011)	WTP for groundwater quality improvement (Groundwater protection from nitrate pollution of human origin requires the development of sewerage systems)	Respondents are willing to pay more than their current water bill (WTP 53.66–54.11 PLN (EUR13.09–13.20)) to secure better water quality.	$57.25-57.72
Rinaudo and Aulong (2014)	CV	The upper Rhine valley alluvial aquifer is located between Germany and France	650 respondents (2006)	WTP for restoring groundwater for potable water WTP for restoring natural groundwater quality (no traces of solvents)	€42/year/household (in 2006 €) over 10 years WTP is €76/year/household (in 2006 €) over 10 years	$64.27 $116.30
Gutrich et al. (2016)	A dynamic ecological economic simulation model	Owens Valley, California	(Model simulations from 2011–2060)	Economic trade-off between pumping groundwater and maintaining a native plant community that provides an ecosystem service of dust suppression	Adaptive management that pumps less water, but high volumes in wet years and low volumes in dry years generates economic rent of $82.6 million (in 2011)	$ 88.13 million
Rupérez-Moreno et al. (2015)*	CV	Boquerón aquifer (Albacete, SE Spain)	240 households of Hellín	WTP to improve ecological status of the water bodies of the aquifer by users WTP by non-users	€ 35.88 per year per inhabitant € 14.86 per year per inhabitant	
Hérivaux and Rinaudo (2016)*	CV	Meuse alluvial aquifer (under the city of Liege, Belgium) The Lower Triassic Sandstone (Lorraine region, in Eastern France)	530 respondents 650 respondents	WTP for groundwater protection WTP for groundwater protection	€ 40 household/year over 10 years € 39 household/year over 10 years	

Note: CV refers to contingent valuation method; CE refers to choice experiments method; * unless otherwise indicated $ = $US. US$1 = AU$1.355, 2016, Reserve Bank Australia
*These studies did not report the survey/study year. Therefore, monetary values were not adjusted for 2016 for those studies.

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. 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. In this section, we will summarize the economic evaluation studies in terms of the main methods employed: averting behaviour studies, contingent valuation studies, choice experiment and hedonic studies. The section concludes with a summary table of all relevant non-market studies identified that are related to water supply (see Table 12).

In addition to the specific summaries of individual papers provided below, specific attention is drawn to Van Houtven et al. (2017), which is a meta-analysis of estimates of household's willingness to pay for improved water supply. The study relies on 171 WTP estimates from 60 stated preference studies and found that predicted WTP values ranged from approximately $3 per month (with a 90% confidence interval (90% CI) of $1.1-$6.1) for modest improvements through to $33.5 per month (90% CI $17.9-$66.0) for more substantial improvements. Other general observations were that households where there was an existing high level of water supply services were willing to pay less.

Averting behavior studies

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 areas of 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 argued that using a combination of different resources and datasets results in the estimation of robust WTP values. The authors also noted that unless careful consideration is given to issues such as heteroscedasticity, estimates will be biased in non-linear models.

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 behaviours 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 or 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) undertook a contingent valuation study in Southern Haiti. Based on a total of 170 completed questionnaires the study found that people would pay 1.7% of their monthly household income to have a public stand post near their homes, and would pay 2.1% of their monthly household income for private connections in their yards.

Griffin et al. (1995) considers responses from the same people before and after an actual intervention, which 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 salinity issues with the local water supply. The first survey was conducted in 1988 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 showed 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 be willing to pay Rs56 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 Rs33 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 (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% 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% of average monthly income.

Using a double bounded dichotomous choice contingent valuation method Tamang and Jana (2017) elicited willingness to pay (WTP) for improved water services in the hill town of Darjeeling in India. Their findings revealed WTP was about INR494.00 (US$1 ~ INR60.00 as of March 2014) per month, which is about 12 times the amount they currently pay.

In a CV study conducted in Bangladesh in 2009 where there were 3,000 respondents Gunatilake and Tachiri (2014) found that households mean WTP for improved water supply was Tk301 per month.

Devoto et al. (2012) found that households in urban Morocco would be willing to pay almost double their current water bills for a private water connections at home, versus US$11 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 (on average they spent nearly 18 hours per month collecting water from public connections). In this study the benefits from newly installed private or public water connections seem to be a function of the time saved.

Using survey data from 387 households in North Central Kenya in 2013, Cook et al. (2016) estimated the cost of coping with poor water supply. The median coping cost per household was estimated at US$20 per month.

Tussupova et al. (2015), using the contingent valuation method survey 168 respondents in 2012 in Pavlodar Region, Kazakhstan to examine household WTP for improved piped water supply services. The mean WTP for the maintenance of the individual piped water system was about 1120-1590 KZT per month per household. For a public standpipe, the mean WTP was found to be about 610-950 KZT per month per household

In Jianjun et al. (2016) the contingent valuation method is used for a case study in Songzi, China to find out how much residents were willing to pay for improving drinking water quality and supply reliability. There were 168 respondents, and the mean WTP was estimated at 16.71 Yuan, which was equal to 0.3% of total household income. The results also showed that more educated respondents, households with higher income, and households with fewer household members are, on average, willing to pay more.

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 for 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.

In another study in Canada, Dupont (2013) reported results from a double bounded contingent valuation survey on people's willingness to pay to avoid summer water use restrictions by using reclaimed wastewater. They found that the mean WTP per household per year was between $142 and $155. The values depended on the scale of the project and expectations on neighbouring compliance with summer water use restrictions.

The WTP of residents in ten districts in California, USA to avoid water shortages was investigated in Koss and Khawaja (2001). Using 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% water shortage once every ten years; and US$16.92 per month to avoid a 50% shortage occurring every twenty years.

Epp and Delavan (2001) investigated 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 found 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.

The contingent valuation study conducted by Poe and Bishop (2001) examined 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:

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

The study found a mean WTP of AU$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 AU$6.3 and AU$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 demonstrated that the estimated WTP from studies can be a reasonable representation of the marginal price of water supplies.

In del Saz-Salazar et al. (2016) the contingent valuation method was study WTP for improving urban water supply infrastructure and reducing leakages in the Guadalquivir River basin in Spain. The study is based on data from 531 residents in 2014, and found that, on average, individuals would be willing to pay an extra charge on their water bill ranging from €8.23 to €9.65 to improve water supply infrastructure.

In Beaumais et al. (2014b) households' WTP for improved water quality was analysed using data collected from a contingent valuation study in 10 OECD countries conducted in 2008. The mean WTP was estimated at US$22.62 per annum

The contingent valuation method can also be used to estimate the value of alternative water supplies. For example, 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. The research is, however, silent on whether or not there is a statistically significant difference in residents' preferences for different drinking water supply sources.

In Laughland et al. (1996) 226 households in Milesburg, Pennsylvania, USA were surveyed. 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% more than their existing water bills, for the provision of safe drinking water to their houses.

Recently, Holguín-Veras et al. (2016) estimated deprivation cost functions using contingent valuation technique which is able to capture the economic value of human suffering from loss of water supply. The study does not, however provide a readily comparable value that can be summarised.

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 AU$47 per year. There was, however, 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 AU$41.51 per household. However, if residents currently face monthly interruptions, the mean WTP to reduce the water supply interruptions to bimonthly is only AU$9.58. Households' WTP to reduce wastewater flow from twice a year to once a year was estimated to be AU$77.85, and for reduced wastewater flow from once per year to once every two years was estimated to be AU$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% 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 US$22 per month in house rent. Finally, Alam and Pattanayak (2009) found that households, 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% and 52% to house prices.

Other studies

A recent study carried out in California using a panel dataset (from 1997-2009) of 37 water utilities, examined the welfare losses due to water supply disruptions based on residential water demand (Buck et al., 2016). According to the study, households are willing to pay US$60-600 to avoid an annual water supply disruption depending on shortage size and location.

Molinos-Senante and Sala-Garrido (2017) used directional distance function approach to estimate monetary value for compensation for interruptions in drinking water supply. They used a balanced panel from the 23 main Chilean water companies over the period of (2010–2014). According to study findings, on average, customers should receive a compensation of €0.135 for each hour of unplanned water supply interruptions.

Table 12: Water supply valuation surveys

Author	Method	Location	Number of respondents	Object	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2016)*
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)	$2.40-$2.72
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)	$2.93
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	$7.94-$13.49
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	$41.40
Blamey et al. (1999)	CE	Canberra, Australia	294	Estimate the WTP for possible water supply options (recycled water for outside use or drinking)	AU$47 annual WTP for the provision of recycled water for outdoor use	$46.20
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	$18.61
Whittington (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	$4.60-$20.80
MacDonald et al. (2005)	CE	Adelaide, Australia	337	Estimate the WTP for improved continuity of water supply	AU$1.10 to AU$4.40 per year for decreased duration of water service interruptions; AU$6.00 to AU$15.40 per year for decreased frequency of interruptions in water services	$1.05-$14.84
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 AU$9.58; two interruptions per year AU$41.51; AU$116 to reduce number of wastewater overflow to one time per year;	$9.20-$114.64

Author	Method	Location	Number of respondents	Object	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2016)*
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	$2.20-$8.57
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-$5.23
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	$3.35-$9.20
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)	$6.79
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	$14.95
Hensher et al. (2006)	CE	Canberra, Australia	416	Estimate households' and businesses' WTP to avoid drought water restrictions	AU$11.95 per year to reduce frequency of restrictions from once every 10 years to once every 20 years; AU$3.98 per year to reduce water restriction from once every 20 years to once every 30 years; AU$1,104 (23% of current water bill) by business respondents to avoid severe restrictions	$3.66-$10.98 for household; $1,011.90 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	Number of respondents	Object	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2016)*
Genius et al. (2008)	CV	Rethymno, Greece	306	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)	$15.13
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	$2.40-$2.83
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	$2.40
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	$0.16-$4.08
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)	$0.41-$0.54
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)	$13.69-$18.40
Cooper et al. (2011)	CV	New South Wales and Victoria, Australia	472	Estimate consumers' WTP to avoid urban water restrictions	$6-117 per year as the median WTP	$6.79-$132.76

Author	Method	Location	Number of respondents	Object	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2016)*
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)	$3.24-$9.62
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)	$8.47
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)	$0.21
Tarfasa and Brouwer (2013)	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	$0.63-$1.88
Beaumais et al. (2014a)	CV	10 OECD countries	10,000 (2008)	Households willingness to pay for better tap water quality	$22.62 per annum	$25.22
Gunatilake and Tachiri (2014)	CV	Khulna city Bangladesh,	3000 (2009)	Mean WTP for improved water supply	Tk301 per month.	$4.95
Tesfaye and Brouwer (2015)	CE	Blue Nile river basin Sudan	200 (2012)	Downstream farmers willingness to pay for upstream and use changes to improve irrigation water supply	MWTP for transboundary cooperation MWTP to increase irrigation frequency (1 extra time per year) - $1.6 per ha MWTP for water efficient sprinkler irrigation - $0.7 per ha	$1.67 $0.73
Tussupova et al. (2015)	CV	Pavlodar Region, Kazakhstan	168 (2012)	Willingness to Pay to Improve piped water supply services	The mean WTP for the maintenance of the individual piped water system was about 1120-1590 KZT per month per household For public standpipe, the mean WTP was about 610-950 KZT per month per household	$7.92-11.24 $4.31- 6.71

Author	Method	Location	Number of respondents	Object	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2016)*
Buck et al. (2016)	Estimation of residential water demand	California	453 Panel data set of 37 California water utilities (1997-2009)	Welfare losses due to water supply disruptions	Household-level willingness to pay to avoid an annual disruption of approximately $60–$600 depending on the shortage size and location.	$67.12-671.23
Cook et al. (2016)	Household survey	Kianjai, North Central Kenya	387 (2013)	The costs of coping with poor water supply	Median total coping costs per month are approximately US$20 per month	$20.61
Giannoccaro et al. (2016)	CV	Guadalquivir and Almanzora River Basins, Southern Spain	241 (2012)	Willingness to participate in water allocation trading	Mean WTP in a normal year was €0.35/m^3/ha Mean WTP in a drought year was € 0.37/m^3/ha	$0.37 $0.39
Jianjun et al. (2016)	CV	Songzi,China	168 (2011)	Willingness to pay for drinking water quality improvements	The mean WTP for the drinking water quality improvement program was estimated to be 16.71 yuan (0.3% of total household income	$17.83
del Saz-Salazar et al. (2016)	CV	Guadalquivir River basin in Spain	531 (2014)	WTP for improving urban water supply infrastructure and reducing leakages	On average individuals would be willing to pay an extra charge on their water bill ranging from €8.23 to €9.65	$11.40-13.37
Tamang and Jana (2017)	CV	Darjeeling, India	(2014)	Willingness to pay for improved water services	WTP was about INR494.00 (US$1 ~ INR60.00 as of March 2014) per month	$ 8.39
Molinos-Senante and Sala-Garrido (2017)	Directional distance function approach	Chilean water industry	A balanced panel from the 23 main Chilean water companies over the period of (2010–2014)	Compensation for interruptions in the drinking water supply	On average, customers should receive a compensation of €0.135 for each hour of unplanned water supply interruptions	$0.19
Van Houtven et al. (2017)	Meta-analysis	A worldwide sample	171 WTP estimates from 60 studies (2008)	WT for improved water access	households are willing to pay between approximately $3 and $30 per month for improvements in water access.	$3.34-33.34

Note: CV refers to contingent valuation method; CE refers to choice experiments method; * unless otherwise indicated $ = $US. US$1 = AU$1.355, 2016, Reserve Bank Australia

Wastewater management

Globally, around 90% 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% of wastewater is recycled for reuse (Dimitriadis, 2005).

Historically, in Australia wastewater systems have been linear one-directional systems where the main objective was to collect wastewater and stormwater and discharge these flows as quickly as possible. The social dimension and potential multi-functional benefits from wastewater re-use was largely ignored. Perraton et al. (2015) identified six major barriers in wastewater reuse: unsupportive institutional and governance arrangement, difficulties in determining the true cost of disposal options, issues of competition (absence of effective market for urban water) and demand, inadequate water quality management, political and policy influence on decision making and perceptions of integrated water supply options (public acceptance and perception of risks).

Economic valuation studies could be useful to overcome these barriers. From the various non-market valuation methods available, the most commonly used method for investigating wastewater questions 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. Specific details are summarised below, with studies that report WTP values detailed in Table 13.

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% reduction of summer water use was about $CA9.26 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).

Balasubramanya et al. (2017) conducted a contingent valuation survey with 1,091 households in 44 villages in the Bhaluka subdistrict of Bangladesh. The study focused on analyses of households' willingness to pay for emptying and transporting sludge in rural districts with high rates of access to latrines. The estimated WTP was BDT507 per household per event. It was then noted that at this level household contributions would cover only 47% of the cost of emptying and transport.

Research conducted in urban Ethiopia using the contingent valuation method found that WTP for a medium level improvement in treatment of liquid waste was US$0.93 per month (Woldemariam et al., 2016). The study was based on 384 residents and was conducted in 2011.

Vásquez and Espaillat (2016) use the contingent valuation method to estimate willing to pay for reliable supplies of safe drinking water. The study was based on 500 respondents in Guatemala, and was conducted in 2012. The median WTP was estimated at 51 quetzals (US$6.50), which would imply an increase in the average water bill of more than 250%.

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 an increase in household water costs of about AU$47. In Western Sydney, based on a survey of 800 households, (Bennett et al., 2016) found that there was a community preference for increased use of recycled water, however, residents first preference was to replace use of potable water for industrial uses with recycled water.

In Greece it was found that irrigators preferred the use of recycled water for perennial horticulture (such as olive gardens) rather than for vegetables, vines or ornamental plants (Petousi et al., 2015). As indicated above the use of recycled water could improve the reliability of services and people would be willing to pay more to ensure it. Further, in Spain, it was found that farmers were willing to pay twice as much as their current irrigation water price for recycled water to ensure water supply reliability through government supply guaranteed programs (Alcon et al., 2014).

Birol and Das (2010) used choice experiments to estimate residents' willingness to pay for improved capacity and technology at a sewage treatment plant in the 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. In another study in Hyderabad, India, Saldías et al. (2017) found farmers were willing to pay US$18.8/ha/year to obtain a treated water. Further, they were willing to pay $US14.7/ha/year to reduce health risk from 'tolerable health risks' to 'reduced health risks'.

Preferences for policy attributes and willingness to pay for water quality improvements under uncertainty was examined by Mullen et al. (2017). The study collected data from 334 respondents in 2010 and found that the annual willingness to pay for a 90% reduction in the probability of failing to meet water quality standards for a 5 year period was between $0 and $300.

Shadow price evaluation method

By using the concept of a 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 resource, 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) and Molinos-Senante et al. (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.

Some studies examined the possibility of using wastewater to produce compost and thereby reducing greenhouse gas emissions (Department of Environment, 2011). A case study in Sydney focussing on a community’s willingness to pay for waste water found that the value is in the order of $0.45-$3.80 per Kl depending on the volume and the end use. For values at the higher end of this range, the impact on the viability of any scheme would be highly significant (Australian Water Recycling Centre of Excellence, 2013).

Table 13: Wastewater valuation surveys

Author	Method	Location	Number of respondents	Study	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2016)
Balasubramanya et al. (2017)	CV	Bhaluka subdistrict in Bangladesh	1,091 households in 44 villages (2011)	Willingness to pay for emptying and transporting sludge in rural districts with high rates of access to latrines	Willing to pay up to BDT 507 for the emptying and transport away of the contents of their pit latrines	$7.41
Dupont (2013)	CV	Canada	1135 (2009)	WTP for supplementing existing water supplies with reclaimed wastewater	Average annual WTP per household was b $142 - $155	$158.86-$173.40
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	$71.82-$129.62
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	$49.65
Hernández-Sancho et al. (2010)	Distance functions	Spain	43 wastewater treatment plants (2004)	Economic valuation of environmental benefits from wastewater treatment processes:	Environmental benefit resulting from wastewater treatment stands at €0.807 per cubic meter.	$1.03
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	$3.45-$12.13
Hagen et al. (2017)	Longitudinal study including CV	Maricopa County, Arizona	380	Socioeconomic benefits of wastewater treatment projects in a desert city	In the short run, mean discount % on $100,000 house located less than 0.25 miles away from the water treatment facility was 32.3%. In the short run, mean discount% on $100,000 house located within 0.5-1 mile was 28.7%. In the long run property values increased significantly due to investment in state-of art technologies at the facility	
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	$18.61

Author	Method	Location	Number of respondents	Study	Mean WTP estimates (per household)*	Adjusted WTP value (value in $US2016)
Mullen et al. (2017)	CE	Gwinnett County, Georgia	334 (2010)	Preferences for policy attributes and willingness to pay for water quality improvements under uncertainty	Depending on the status quo, an annual willingness to pay for a 90% reduction in the probability of failing to meet water quality standards between $0 and $300 for a 5 year period.	$0- 330.20
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	$29.90
Woldemariam et al. (2016)	CV	Urban Ethiopia	384 (2011)	Residents' willingness to pay for improved liquid waste treatment	WTP for the medium improvement was found to be ETB15.53 (US$0.93) per month.	$0.99
Vásquez and Espaillat (2016)	CV	Guatemala	500 (2012)	Willingness to pay for reliable supplies of safe drinking water	WTP significant increase in their water bills for uninterrupted supplies of safe drinking water. The median WTP is estimated at 51 quetzals (US$6.50), which would imply an increase in the average water bill of more than 250%.	$6.79

Conclusions

Adopting water sensitive systems and practices has the potential to provide significant benefits in terms of improving liveability, providing amenity benefits, improving water quality, tackling climate change, reducing flood risk, protecting groundwater, securing water supply and supporting the environment and ecosystems. The economic value of these benefits are often captured through various non-market valuation methods.

Our review provides an overview of the available information on monetized values of these services. The information could be used to get an understanding of the extent of benefits. However, to use them for any particular context and locations these estimates need to be properly adjusted using appropriate benefit transfer techniques. Estimation methods, uncertainty in estimated values and scope of evaluation would need to be properly considered during adjustment.

We found that 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.

Finally, even though there is widespread recognition of multi-functional benefits of water sensitive systems and practices, there is a lack of examples where non-market values have been successfully integrated in an economic analysis of water sensitive projects. Therefore, proper policies and guidelines are needed to encourage regulators and utilities to make decisions that consider the full range of social and economic costs and benefits.

Appendix A: Non-market valuation methods

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 utilization 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 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 behavior 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 non-market 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 14). 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 14: Illustrative example of a choice set used in the CRC for Water Sensitive Cities wastewater buffer zone survey

There are 4 different land use categories below to consider in each option	Option 1 (Current situation)	Option 2	Option 3
Nature conservation areas 	No change: 30%	10%	50%
Commercial / industrial areas 	No change: 30%	20%	10%
Agricultural areas 	No change: 30%	50%	30%
Sporting and recreation areas 	No change: 10%	20%	10%
There is an increase in your service provider's bill (e.g. water bill) for each option (in $ per **quarter**)	$0	$21	$39
Most preferred option	□	□	□
Least preferred option	□	□	□

Source: Zhang et al. (2015a)

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.

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 reviewed 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 localized 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%, and the transfer error can be more than 200% 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 et al., 2007). Meta-regression analysis in particular can be used to synthesise existing research findings when there are many varying study attributes (Glass et al., 1984). 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.

References

ABDALLA, C. W. 1990. Measuring economic losses from ground water contamination: An investigation of household avoidance costs. *JAWRA Journal of the American Water Resources Association,* 26**,** 451-463.

AECOM 2017. Green infrastrucutre: A vitla step to brilliant Australian cities.

AKBARI, H. 2002. Shade trees reduce building energy use and CO2 emissions from power plants. *Environmental Pollution,* 116**,** S119-S126.

AKRAM, A. A. & OLMSTEAD, S. M. 2011. The value of household water service quality in Lahore, Pakistan. *Environmental and Resource Economics,* 49**,** 173-198.

ALAM, M. & PATTANAYAK, S. Coping costs of unsafe and unreliable drinking water: the case of slums in Dhaka, Bangladesh. 17th Annual Conference of European Association of Environmental and Resource Economists in Amsterdam, the Netherlands, 2009. 24-27.

ALBERINI, A. & KAHN, J. 2006. *Handbook on contingent valuation*, Edward Elgar.

ALCOCK, I., WHITE, M. P., WHEELER, B. W., FLEMING, L. E. & DEPLEDGE, M. H. 2014. Longitudinal Effects on Mental Health of Moving to Greener and Less Green Urban Areas. *Environmental Science & Technology,* 48**,** 1247-1255.

ALCON, F., TAPSUWAN, S., BROUWER, R. & DE MIGUEL, M. D. 2014. Adoption of irrigation water policies to guarantee water supply: A choice experiment. *Environmental Science & Policy,* 44**,** 226-236.

ALTAF, M. A., WHITTINGTON, D., JAMAL, H. & SMITH, V. K. 1993. Rethinking rural water supply policy in the Punjab, Pakistan. *Water Resources Research,* 29**,** 1943-1954.

ALVAREZ, S., ASCI, S. & VOROTNIKOVA, E. 2016. Valuing the potential benefits of water quality improvements in watersheds affected by non-point source pollution. *Water,* 8**,** 112.

AMBREY, C. & FLEMING, C. 2014. Public Greenspace and Life Satisfaction in Urban Australia. *Urban Studies,* 51**,** 1290-1321.

ANDERSON, S. T. & WEST, S. E. 2006. Open space, residential property values, and spatial context. *Regional Science and Urban Economics,* 36**,** 773-789.

ANDREWS, B., FERRINI, S. & BATEMAN, I. 2017. Good parks – bad parks: the influence of perceptions of location on WTP and preference motives for urban parks. *Journal of Environmental Economics and Policy***,** 1-21.

AULONG, S. & RINAUDO, J.-D. Assessing the benefits of different groundwater protection levels: results and lessons learnt from a contingent valuation survey in the Upper Rhine valley aquifer, France. XIVth IWRA World Water Congress. IWRA. Montpellier, 2008.

AUSTRALIAN WATER RECYCLING CENTRE OF EXCELLENCE 2013. The Economic Viability of Recycled Water Schemes for Non-potable Use.

AWAD, I. M. 2012. Using econometric analysis of willingness-to-pay to investigate economic efficiency and equity of domestic water services in the West Bank. *The Journal of Socio-Economics,* 41**,** 485-494.

BADLAND, H., WHITZMAN, C., LOWE, M., DAVERN, M., AYE, L., BUTTERWORTH, I., HES, D. & GILES-CORTI, B. 2014. Urban liveability: Emerging lessons from Australia for exploring the potential for indicators to measure the social determinants of health. *Social Science & Medicine,* 111**,** 64-73.

BALASUBRAMANYA, S., EVANS, B., HARDY, R., AHMED, R., HABIB, A., ASAD, N., RAHMAN, M., HASAN, M., DEY, D. & FLETCHER, L. 2017. Towards sustainable sanitation management: Establishing the costs and willingness to pay for emptying and transporting sludge in rural districts with high rates of access to latrines. *PloS one,* 12**,** e0171735.

BARTON, J., HINE, R. & PRETTY, J. 2009. The health benefits of walking in greenspaces of high natural and heritage value. *Journal of Integrative Environmental Sciences,* 6**,** 261-278.

BARTON, K. 1978. The other water pollution. *Environment: Science and Policy for Sustainable Development,* 20**,** 12-20.

BARTOSOVA, A., CLARK, D., NOVOTNY, V. & TAYLOR, K. S. 2000. Using GIS to evaluate the effects of flood risk on residential property values. *Environmental Problem Solving with Geographical Information Systems: A National Conference.* Cincinnati, OH: U.S: Environmental Protection Agency.

BATEMAN, I. J., CARSON, R. T., DAY, B., HANEMANN, M., HANLEY, N., HETT, T., JONES-LEE, M. & LOOMES, G. 2002. Economic valuation with stated preference techniques. *Books*.

BATEMAN, I. J., LANGFORD, I. H. & GRAHAM, A. 1995. *A survey of non-users willingness to pay to prevent saline flooding in the Norfolk broads*, Centre for Social and Economic Research on the Global Environment.

BEAUMAIS, O., BRIAND, A., MILLOCK, K. & NAUGES, C. 2014a. What are households willing to pay for better tap water quality? A cross-country valuation study.

BEAUMAIS, O., MILLOCK, K. & NAUGES, C. 2014b. What are households willing to pay for better tap water quality? a cross-country valuation study.

BECKER, N., INBAR, M., BAHAT, O., CHORESH, Y., BEN-NOON, G. & YAFFE, O. 2005. Estimating the economic value of viewing griffon vultures Gyps fulvus: A travel cost model study at Gamla Nature Reserve, Israel. *Oryx,* 39**,** 429-434.

BEDIMO-RUNG, A. L., MOWEN, A. J. & COHEN, D. A. 2005. The significance of parks to physical activity and public health: A conceptual model. *American Journal of Preventive Medicine,* 28**,** 159-168.

BEHAILU, S., KUME, A. & DESALEGN, B. 2012. Household's willingness to pay for improved water service: a case study in Shebedino District, Southern Ethiopia. *Water and Environment Journal,* 26**,** 429-434.

BENNETT, J. & BLAMEY, R. 2001. *The choice modelling approach to environmental valuation*, Edward Elgar Publishing.

BENNETT, J., DUMSDAY, R., HOWELL, G., LLOYD, C., STURGESS, N. & VAN RAALTE, L. 2008. The economic value of improved environmental health in Victorian rivers. *Australasian Journal of Environmental Management,* 15**,** 138-148.

BENNETT, J., MCNAIR, B. & CHEESMAN, J. 2016. Community preferences for recycled water in Sydney. *Australasian Journal of Environmental Management,* 23**,** 51-66.

BENNETT, J. & MORRISON, M. 2001. Valuing the environmental attributes of NSW rivers.

BERNATH, K. & ROSCHEWITZ, A. 2008. Recreational benefits of urban forests: Explaining visitors' willingness to pay in the context of the theory of planned behavior. *Journal of Environmental Management,* 89**,** 155-166.

BEYER, K., KALTENBACH, A., SZABO, A., BOGAR, S., NIETO, F. & MALECKI, K. 2014. Exposure to Neighborhood Green Space and Mental Health: Evidence from the Survey of the Health of Wisconsin. *International Journal of Environmental Research and Public Health,* 11**,** 3453.

BIN, O. & POLASKY, S. 2004. Effects of flood hazards on property values: evidence before and after Hurricane Floyd. *Land Economics,* 80**,** 490-500.

BIROL, E. & DAS, S. 2010. Estimating the value of improved wastewater treatment: The case of River Ganga, India. *Journal of environmental management,* 91**,** 2163-2171.

BLAMEY, R., GORDON, J. & CHAPMAN, R. 1999. Choice modelling: assessing the environmental values of water supply options. *Australian Journal of Agricultural and Resource Economics,* 43**,** 337-357.

BLOMQUIST, G. 1988. Valuing urban lakeview amenities using implicit and contingent markets. *Urban studies,* 25**,** 333-340.

BLONG, R. 2003. A new damage index. *Natural hazards,* 30**,** 1-23.

BOURCIER, D. R., HINDIN, E. & COOK, J. C. 1980. Titanium and tungsten in highway runoff at Pullman, Washington. *International Journal of Environmental Studies,* 15**,** 145-149.

BOWLER, D. E., BUYUNG-ALI, L., KNIGHT, T. M. & PULLIN, A. S. 2010. Urban greening to cool towns and cities: A systematic review of the empirical evidence. *Landscape and Urban Planning,* 97**,** 147-155.

BOYLE, K. J. & BERGSTROM, J. C. 1992. Benefit transfer studies: myths, pragmatism, and idealism. *Water Resources Research,* 28**,** 657-663.

BRACK, C. L. 2002. Pollution mitigation and carbon sequestration by an urban forest. *Environmental Pollution,* 116, Supplement 1**,** S195-S200.

BRANDER, L. M. & KOETSE, M. J. 2011. The value of urban open space: Meta-analyses of contingent valuation and hedonic pricing results. *Journal of Environmental Management,* 92**,** 2763-2773.

BRENT, D. A., GANGADHARAN, L., LASSITER, A., LEROUX, A. & RASCHKY, P. A. 2016. Valuing Environmental Services Provided by LocalStormwater Management. Monash University, Department of Economics.

BRISCOE, J., DE CASTRO, P. F., GRIFFIN, C., NORTH, J. & OLSEN, O. 1990. Toward equitable and sustainable rural water supplies: a contingent valuation study in Brazil. *The World Bank Economic Review,* 4**,** 115-134.

BROUWER, R. 2000. Environmental value transfer: state of the art and future prospects. *Ecological economics,* 32**,** 137-152.

BROUWER, R. & BATEMAN, I. J. 2005. Temporal stability and transferability of models of willingness to pay for flood control and wetland conservation. *Water Resources Research,* 41.

BROWN, R. R. & FARRELLY, M. A. 2009. Delivering sustainable urban water management: a review of the hurdles we face. *Water Science and Technology,* 59**,** 839-846.

BROWN, T. C. & DANIEL, T. C. 1991. Landscape aesthetics of riparian environments: relationship of flow quantity to scenic quality along a wild and scenic river. *Water Resources Research,* 27**,** 1787-1795.

BUCK, S., AUFFHAMMER, M., HAMILTON, S. & SUNDING, D. 2016. Measuring Welfare Losses from Urban Water Supply Disruptions. *Journal of the Association of Environmental and Resource Economists,* 3**,** 743-778.

CARSON, R. T., FLORES, N. E. & MEADE, N. F. 2001. Contingent valuation: controversies and evidence. *Environmental and resource economics,* 19**,** 173-210.

CARSON, R. T. & MITCHELL, R. C. 1993. The Value of clean water: The public's willingness to pay for boatable, fishable, and swimmable quality water. *Water Resources Research,* 29**,** 2445-2454.

CASEY, J. F., KAHN, J. R. & RIVAS, A. 2006. Willingness to pay for improved water service in Manaus, Amazonas, Brazil. *Ecological Economics,* 58**,** 365-372.

CHEN, W. Y. & JIM, C. Y. 2008. Cost-benefit analysis of the leisure value of urban greening in the new Chinese city of Zhuhai. *Cities,* 25**,** 298-309.

CHIVERS, J. F. 2001. *Flood Risk, Property Values and Information Market Failure.* Master's, University of Colorado.

CHO, S.-H., POUDYAL, N. C. & ROBERTS, R. K. 2008. Spatial analysis of the amenity value of green open space. *Ecological Economics,* 66**,** 403-416.

CHONG, H. & SUNDING, D. 2006. Water markets and trading. *Annu. Rev. Environ. Resour.,* 31**,** 239-264.

CLEMENTS-CROOME, D. 2015. Creative and productive workplaces: a review. *Intelligent Buildings International,* 7**,** 164-183.

COMMONWEALTH OF AUSTRALIA 2015. State of Australian Cities 2014-2015: Progress in Australian Regions. Commonwealth Department of Infrastructure and Regional Development, Commonwealth of Australia.

CONNELLY, N. A., BROWN, T. L. & BROWN, J. W. 2007. Measuring the net economic value of recreational boating as water levels fluctuate. *JAWRA Journal of the American Water Resources Association,* 43**,** 1016-1023.

COOK, J., KIMUYU, P. & WHITTINGTON, D. 2016. The costs of coping with poor water supply in rural Kenya. *Water Resources Research,* 52**,** 841-859.

COOMBES, E., JONES, A. P. & HILLSDON, M. 2010. The relationship of physical activity and overweight to objectively measured green space accessibility and use. *Social science & medicine,* 70**,** 816-822.

COOPER, B., BURTON, M. & CRASE, L. 2011. Urban water restrictions: Attitudes and avoidance. *Water Resources Research,* 47.

COUTTS, A. M., TAPPER, N. J., BERINGER, J., LOUGHNAN, M. & DEMUZERE, M. 2013. Watering our cities: the capacity for water sensitive urban design to support urban cooling and improve human thermal comfort in the Australian context. *Progress in Physical Geography,* 37**,** 2-28.

CRASE, L. & GILLESPIE, R. 2008. The impact of water quality and water level on the recreation values of Lake Hume. *Australasian Journal of Environmental Management,* 15**,** 21-29.

DAIGGER, G. T. & CRAWFORD, G. V. 2007. Enhancing water system security and sustainability by incorporating centralized and decentralized water reclamation and reuse into urban water management systems. *Journal of Environmental Engineering and Management,* 17**,** 1.

DAMIGOS, D. & KALIAMPAKOS, D. 2003. Assessing the benefits of reclaiming urban quarries: a CVM analysis. *Landscape and urban planning,* 64**,** 249-258.

DANIEL, V. E., FLORAX, R. J. & RIETVELD, P. 2009. Flooding risk and housing values: An economic assessment of environmental hazard. *Ecological Economics,* 69**,** 355-365.

DAVIES, Z. G., EDMONDSON, J. L., HEINEMEYER, A., LEAKE, J. R. & GASTON, K. J. 2011. Mapping an urban ecosystem service: quantifying above-ground carbon storage at a city-wide scale. *Journal of Applied Ecology,* 48**,** 1125-1134.

DE KONING, K., FILATOVA, T. & BIN, O. 2017. Bridging the Gap Between Revealed and Stated Preferences in Flood-prone Housing Markets. *Ecological Economics,* 136**,** 1-13.

DEL SAZ-SALAZAR, S., GARCÍA-RUBIO, M. A., GONZÁLEZ-GÓMEZ, F. & PICAZO-TADEO, A. J. 2016. Managing water resources under conditions of scarcity: on consumers' willingness to pay for improving water supply infrastructure. *Water resources management,* 30**,** 1723-1738.

DEL SAZ-SALAZAR, S. & RAUSELL-KÖSTER, P. 2008. A double-hurdle model of urban green areas valuation: dealing with zero responses. *Landscape and urban planning,* 84**,** 241-251.

DEL SAZ SALAZAR, S. & MENENDEZ, L. G. 2007. Estimating the non-market benefits of an urban park: Does proximity matter? *Land Use Policy,* 24**,** 296-305.

DELOITTE ACCESS ECONOMICS PTY LTD 2013. Economic Value of Groundwater in Australia. National Centre for Groundwater Research and Training.

DEMUZERE, M., ORRU, K., HEIDRICH, O., OLAZABAL, E., GENELETTI, D., ORRU, H., BHAVE, A. G., MITTAL, N., FELIU, E. & FAEHNLE, M. 2014. Mitigating and adapting to climate change: Multi-functional and multi-scale assessment of green urban infrastructure. *Journal of Environmental Management,* 146**,** 107-115.

DEPARTMENT OF ENVIRONMENT AND CONSERVATION 2006. Managing urban stormwater: harvesting and reuse (Case studies). Sydney: Department of Environment and Conservation NSW.

DEPARTMENT OF ENVIRONMENT, C. C. A. W. N. 2011. Short report: The benefits of using compost for mitigating climate change.

DERKZEN, M. L., VAN TEEFFELEN, A. J. A. & VERBURG, P. H. 2017. Green infrastructure for urban climate adaptation: How do residents' views on climate impacts and green infrastructure shape adaptation preferences? *Landscape and Urban Planning,* 157**,** 106-130.

DEVOTO, F., DUFLO, E., DUPAS, P., PARIENTÉ, W. & PONS, V. 2012. Happiness on tap: piped water adoption in urban Morocco. *American Economic Journal: Economic Policy,* 4**,** 68-99.

DIMITRIADIS, S. 2005. Issues encountered in advancing Australia's water recycling schemes. *Research Brief.* Department of Parliamentary Services, Parliament of Australia, 2005–2006.

DONOVAN, G. H. & BUTRY, D. T. 2009. The value of shade: Estimating the effect of urban trees on summertime electricity use. *Energy and Buildings,* 41**,** 662-668.
DONOVAN, G. H. & BUTRY, D. T. 2010. Trees in the city: Valuing street trees in Portland, Oregon. *Landscape and Urban Planning,* 94**,** 77-83.
DONOVAN, G. H. & BUTRY, D. T. 2011. The effect of urban trees on the rental price of single-family homes in Portland, Oregon. *Urban Forestry & Urban Greening,* 10**,** 163-168.
DONOVAN, G. H., MICHAEL, Y. L., BUTRY, D. T., SULLIVAN, A. D. & CHASE, J. M. 2011. Urban trees and the risk of poor birth outcomes. *Health & Place,* 17**,** 390-393.
DUMENU, W. K. 2013. What are we missing? Economic value of an urban forest in Ghana. *Ecosystem Services,* 5**,** E137-E142.
DUPONT, D. P. Reclaimed wastewater and the wtp to avoid summer water restrictions: Incorporating endogenous free-riding beliefs. 85th annual conference of the Agricultural Economics Society, Warwick University. Retrieved from http://ageconsearch. umn. edu/bitstream/108778/2/54dupont. pdf, 2011.
DUPONT, D. P. 2013. Water use restrictions or wastewater recycling? A Canadian willingness to pay study for reclaimed wastewater. *Water Resources and Economics,* 1**,** 61-74.
EDWARDS, S. F. 1988. Option prices for groundwater protection. *Journal of Environmental Economics and Management,* 15**,** 475-487.
EPP, D. J. & DELAVAN, W. 2001. Measuring the value of protecting ground water quality from nitrate contamination in southeastern Pennsylvania. *The Economic Value of Water Quality. Cheltenham, UK: Edward Elgar*.
ESCOBEDO, F., VARELA, S., ZHAO, M., WAGNER, J. E. & ZIPPERER, W. 2010. Analyzing the efficacy of subtropical urban forests in offsetting carbon emissions from cities. *Environmental Science & Policy,* 13**,** 362-372.
FARBER, S. 1992. The economic cost of residual environmental risk: a case study of Louisiana. *Journal of Environmental Management,* 36**,** 1-16.
FARR, M., STOECKL, N. & ALAM BEG, R. 2014. The non-consumptive (tourism) 'value' of marine species in the Northern section of the Great Barrier Reef. *Marine Policy,* 43**,** 89-103.
FEMA 2012a. Engineering principles and practices of retrofitting flood-prone residential structures (Third Edition). Federal Emergency Management Agency.
FEMA 2012b. *Engineering Principles and Practices of Retrofitting Flood-prone Residential Structures, Third Edition*, Springer-Verlag.
FLEMING, C. M. & COOK, A. 2008. The recreational value of Lake McKenzie, Fraser Island: An application of the travel cost method. *Tourism Management,* 29**,** 1197-1205.
FRANCIS, J., WOOD, L. J., KNUIMAN, M. & GILES-CORTI, B. 2012. Quality or quantity? Exploring the relationship between Public Open Space attributes and mental health in Perth, Western Australia. *Social Science & Medicine,* 74**,** 1570-1577.
FRASER, R. & SPENCER, G. 1998. The Value of an Ocean View: an Example of Hedonic Property Amenity Valuation. *Australian Geographical Studies,* 36**,** 94-98.
FUJITA, Y., FUJII, A., FURUKAWA, S. & OGAWA, T. 2005. Estimation of willingness-to-pay (WTP) for water and sanitation services through contingent valuation method (CVM): A case study in Iquitos City, The Republic of Peru. *Japan Bank International Cooperation Institute Review* 11**,** 59-87.
GE, J., KLING, C. L. & HERRIGES, J. A. 2013. How much is clean water worth? Valuing water quality improvement using a meta analysis. *Economics Working Papers (2002–2016).* Iowa: Iowa State University.
GENIUS, M., HATZAKI, E., KOUROMICHELAKI, E. M., KOUVAKIS, G., NIKIFORAKI, S. & TSAGARAKIS, K. P. 2008. Evaluating consumers' willingness to pay for improved potable water quality and quantity. *Water Resources Management,* 22**,** 1825-1834.
GENIUS, M., MANIOUDAKI, M., MOKAS, E., PANTAGAKIS, E., TAMPAKAKIS, D. & TSAGARAKIS, K. P. 2005. Estimation of willingness to pay for wastewater treatment. *Water Science and Technology: Water Supply,* 5**,** 105-113.
GENIUS, M. & TSAGARAKIS, K. P. 2006. Water shortages and implied water quality: A contingent valuation study. *Water Resources Research,* 42.
GIANNOCCARO, G., CASTILLO, M. & BERBEL, J. 2016. Factors influencing farmers' willingness to participate in water allocation trading. A case study in southern Spain. *Spanish Journal of Agricultural Research,* 14**,** 0101.
GIDLÖF-GUNNARSSON, A. & ÖHRSTRÖM, E. 2007. Noise and well-being in urban residential environments: The potential role of perceived availability to nearby green areas. *Landscape and Urban Planning,* 83**,** 115-126.
GILES-CORTI, B., BROOMHALL, M. H., KNUIMAN, M., COLLINS, C., DOUGLAS, K., NG, K., LANGE, A. & DONOVAN, R. J. 2005. Increasing walking: How important is distance to, attractiveness, and size of public open space? *American Journal of Preventive Medicine,* 28**,** 169-176.
GILLESPIE, R. & BENNETT, J. 1999. Using Contingent Valuation to Estimate Environmental Improvements Associated with Wastewater Treatment. *Australasian Journal of Environmental Management,* 6**,** 14-20.

GLASS, G. V., MACGAW, B. & SMITH, M. L. 1984. *Meta-analysis in social research*, Sage Beverly Hills, CA.

GOBER, P., BRAZEL, A., QUAY, R., MYINT, S., GROSSMAN-CLARKE, S., MILLER, A. & ROSSI, S. 2009. Using watered landscapes to manipulate urban heat island effects: how much water will it take to cool Phoenix? *Journal of the American Planning Association,* 76**,** 109-121.

GODFREY, S., LABHASETWAR, P. & WATE, S. 2009. Greywater reuse in residential schools in Madhya Pradesh, India—A case study of cost–benefit analysis. *Resources, Conservation and Recycling,* 53**,** 287-293.

GORDON, J., CHAPMAN, R. & BLAMEY, R. 2001. Assessing the options for the Canberra water supply: an application of choice modelling. *In:* BENNETT, J. & BLAMEY, R. (eds.) *The choice modelling approach to environmental valuation.* Cheltenham: Edward Elgar, .

GRIFFIN, C. C., BRISCOE, J., SINGH, B., RAMASUBBAN, R. & BHATIA, R. 1995. Contingent Valuation and Actual Behavior: Predicting Connections to New Water Systems in the State of Kerala, India. *The World Bank Economic Review,* 9**,** 373-395.

GUHATHAKURTA, S. & GOBER, P. 2007. The impact of the Phoenix urban heat island on residential water use. *Journal of the American Planning Association,* 73**,** 317-329.

GUNATILAKE, H. & TACHIRI, M. 2014. Willingness to Pay and Inclusive Tariff Designs for Improved Water Supply Services in Urban Bangladesh. *Journal of Sustainable Development,* 7**,** 212.

GUNAWARDENA, K. R., WELLS, M. J. & KERSHAW, T. 2017. Utilising green and bluespace to mitigate urban heat island intensity. *Science of The Total Environment,* 584–585**,** 1040-1055.

GUTRICH, J. J., GIGLIELLO, K., GARDNER, K. V. & ELMORE, A. J. 2016. Economic returns of groundwater management sustaining an ecosystem service of dust suppression by alkali meadow in Owens Valley, California. *Ecological Economics,* 121**,** 1-11.

HAGEN, B., PIJAWKA, D., PRAKASH, M. & SHARMA, S. 2017. Longitudinal analysis of ecosystem services' socioeconomic benefits: Wastewater treatment projects in a desert city. *Ecosystem Services,* 23**,** 209-217.

HALL, M. R. 2012. *The Cost of Pollution: Supporting Cost-effective Options, Evaluation and Pollution Reduction*, Urban Water Security Research Alliance Brisbane, Australia.

HARRINGTON, N. & COOK, P. 2014. Groundwater in Australia. Australia: National Centre for Groundwater Research and Training.

HARRISON, D., T. SMERSH, G. & SCHWARTZ, A. 2001. Environmental determinants of housing prices: the impact of flood zone status. *Journal of Real Estate Research,* 21**,** 3-20.

HASLER, B., LUNDHEDE, T., MARTINSEN, L., NEYE, S. & SCHOU, J. 2005. Valuation of groundwater protection versus water treatment in Denmark by choice experiments and contingent valuation. Denmark: National Environmental Research Institute (NERI), Ministry of the Environment, Denmark.

HAYNES, K. E. & GEORGIANNA, T. D. 1989. Risk assessment of water allocation and pollution treatment policies in a regional economy: Reliability, vulnerability and resiliency in the Yellowstone Basin of Montana. *Computers, environment and urban systems,* 13**,** 75-94.

HEERWAGEN, J. 2000. Green buildings, organizational success and occupant productivity. *Building Research & Information,* 28**,** 353-367.

HEKKERT, P. & WIERINGEN, P. C. W. V. 1996. Beauty in the Eye of Expert and Nonexpert Beholders: A Study in the Appraisal of Art. *The American Journal of Psychology,* 109**,** 389.

HENSHER, D., SHORE, N. & TRAIN, K. 2005. Households' willingness to pay for water service attributes. *Environmental and Resource Economics,* 32**,** 509-531.

HENSHER, D., SHORE, N. & TRAIN, K. 2006. Water supply security and willingness to pay to avoid drought restrictions. *Economic Record,* 82**,** 56-66.

HÉRIVAUX, C. & RINAUDO, J.-D. 2016. Integrated Assessment of Economic Benefits of Groundwater Improvement with Contingent Valuation. *In:* JAKEMAN, A. J., BARRETEAU, O., HUNT, R. J., RINAUDO, J.-D. & ROSS, A. (eds.) *Integrated Groundwater Management: Concepts, Approaches and Challenges.* Cham: Springer International Publishing.

HERNÁNDEZ-SANCHO, F., MOLINOS-SENANTE, M. & SALA-GARRIDO, R. 2010. Economic valuation of environmental benefits from wastewater treatment processes: An empirical approach for Spain. *Science of The Total Environment,* 408**,** 953-957.

HILLSDON, M., PANTER, J., FOSTER, C. & JONES, A. 2006. The relationship between access and quality of urban green space with population physical activity. *Public Health,* 120**,** 1127-1132.

HOEHN, J. P. & KRIEGER, D. J. 2000. An Economic Analysis of Water and Wastewater Investments in Cairo, Egypt. *Evaluation Review,* 24**,** 579-608.

HOFFMAN, E., LATIMER, J., HUNT, C., MILLS, G. & QUINN, J. 1985. Stormwater runoff from highways. *Water, Air, and Soil Pollution,* 25.

HOLGUÍN-VERAS, J., AMAYA-LEAL, J., CANTILLO, V., VAN WASSENHOVE, L. N., AROS-VERA, F. & JALLER, M. 2016. Econometric estimation of deprivation cost functions: A contingent valuation experiment. *Journal of Operations Management,* 45**,** 44-56.

HONG, W. & GUO, R. 2017. Indicators for quantitative evaluation of the social services function of urban greenbelt systems: A case study of shenzhen, China. *Ecological Indicators,* 75**,** 259-267.

HOSTETLER, M., ALLEN, W. & MEURK, C. 2011. Conserving urban biodiversity? Creating green infrastructure is only the first step. *Landscape and Urban Planning,* 100**,** 369-371.

HOWE, C. W., SMITH, M. G., BENNETT, L., BRENDECKE, C. M., FLACK, J. E., HAMM, R. M., MANN, R., ROZAKLIS, L. & WUNDERLICH, K. 1994. The Value of Water Supply Reliability in Urban Water Systems. *Journal of Environmental Economics and Management,* 26**,** 19-30.

HUNGATE, B. A., BARBIER, E. B., ANDO, A. W., MARKS, S. P., REICH, P. B., VAN GESTEL, N., TILMAN, D., KNOPS, J. M. H., HOOPER, D. U., BUTTERFIELD, B. J. & CARDINALE, B. J. 2017. The economic value of grassland species for carbon storage. *Science Advances,* 3.

HURLIMANN, A. C. 2009. Water supply in regional Victoria Australia: A review of the water cartage industry and willingness to pay for recycled water. *Resources, Conservation and Recycling,* 53**,** 262-268.

IFTEKHAR, M. S. & FOGARTY, J. 2017. Impact of water allocation strategies to manage groundwater resources in Western Australia: Equity and efficiency considerations. *Journal of Hydrology,* 548**,** 145-156.

IWMI 2010. Wastewater use in agriculture: not only an issue where water is scarce! : International Water Management Institute.

JIANJUN, J., WENYU, W., YING, F. & XIAOMIN, W. 2016. Measuring the willingness to pay for drinking water quality improvements: results of a contingent valuation survey in Songzi, China. *Journal of water and health,* 14**,** 504-512.

JIM, C. Y. & CHEN, W. Y. 2006a. Impacts of urban environmental elements on residential housing prices in Guangzhou (China). *Landscape and Urban Planning,* 78**,** 422-434.

JIM, C. Y. & CHEN, W. Y. 2006b. Recreation-amenity use and contingent valuation of urban greenspaces in Guangzhou, China. *Landscape and Urban Planning,* 75**,** 81-96.

JIM, C. Y. & CHEN, W. Y. 2006c. Recreation–amenity use and contingent valuation of urban greenspaces in Guangzhou, China. *Landscape and Urban Planning,* 75**,** 81-96.

JIM, C. Y. & CHEN, W. Y. 2008. Assessing the ecosystem service of air pollutant removal by urban trees in Guangzhou (China). *Journal of Environmental Management,* 88**,** 665-676.

JIM, C. Y. & CHEN, W. Y. 2009. Value of scenic views: Hedonic assessment of private housing in Hong Kong. *Landscape and Urban Planning,* 91**,** 226-234.

JIM, C. Y. & CHEN, W. Y. 2010. External effects of neighbourhood parks and landscape elements on high-rise residential value. *Land Use Policy,* 27**,** 662-670.

JOHNSTON, R. J., SEGERSON, K., SCHULTZ, E. T., BESEDIN, E. Y. & RAMACHANDRAN, M. 2011. Indices of biotic integrity in stated preference valuation of aquatic ecosystem services. *Ecological Economics,* 70**,** 1946-1956.

JUN, M.-J. & KIM, H.-J. 2017. Measuring the effect of greenbelt proximity on apartment rents in Seoul. *Cities,* 62**,** 10-22.

KERR, G. N., SHARP, B. M. & WHITE, P. Non-marketed impacts of ground water extraction. Australian Agricultural and Resource Economics Society Conference, Adelaide, January, 2001.

KIM, D.-H., AHN, B.-I. & KIM, E.-G. 2016. Metropolitan Residents’ Preferences and Willingness to Pay for a Life Zone Forest for Mitigating Heat Island Effects during Summer Season in Korea. *Sustainability,* 8**,** 1155.

KIM, J., PARK, J., YOON, D. K. & CHO, G.-H. 2017. Amenity or hazard? The effects of landslide hazard on property value in Woomyeon Nature Park area, Korea. *Landscape and Urban Planning,* 157**,** 523-531.

KLEMICK, H., GRIFFITHS, C., GUIGNET, D. & WALSH, P. 2016. Improving Water Quality in an Iconic Estuary: An Internal Meta-analysis of Property Value Impacts Around the Chesapeake Bay. *Environmental and Resource Economics***,** 1-28.

KO, J.-Y., DAY, J. W., LANE, R. R. & DAY, J. N. 2004. A comparative evaluation of money-based and energy-based cost–benefit analyses of tertiary municipal wastewater treatment using forested wetlands vs. sand filtration in Louisiana. *Ecological Economics,* 49**,** 331-347.

KOMIVES, K. 2003. *Infrastructure, property values, and housing choice: an application of property value models in the developing country context.* PhD, University of North Carolina.

KONIJNENDIJK, C. C., ANNERSTEDT, M., NIELSEN, A. B. & MARUTHAVEERAN, S. 2013. Benefits of urban parks: a systematic review. A report for IPFRA. IFPRA.

KONTOGIANNI, A., LANGFORD, I. H., PAPANDREOU, A. & SKOURTOS, M. S. 2003. Social preferences for improving water quality: An economic analysis of benefits from wastewater treatment. *Water Resources Management,* 17**,** 317-336.

KOSS, P. & KHAWAJA, M. S. 2001. The value of water supply reliability in California:: a contingent valuation study. *Water Policy,* 3**,** 165-174.

KOTCHEN, M., KALLAOS, J., WHEELER, K., WONG, C. & ZAHLLER, M. 2009. Pharmaceuticals in wastewater: Behavior, preferences, and willingness to pay for a disposal program. *Journal of Environmental Management,* 90**,** 1476-1482.

KOUNTOURIS, Y., GODYN, I. & SAUER, J. 2014. Non-market valuation supporting water management: the case study in Czestochowa, Poland. *Hydrology and Earth System Sciences Discussions,* 11**,** 7169-7188.

KRAGT, M. E. & BENNETT, J. W. Using choice experiments to value river and estuary health in Tasmania with individual preference heterogeneity. 2009.

LAFORTEZZA, R., CARRUS, G., SANESI, G. & DAVIES, C. 2009. Benefits and well-being perceived by people visiting green spaces in periods of heat stress. *Urban Forestry & Urban Greening,* 8**,** 97-108.

LATINOPOULOS, D., MALLIOS, Z. & LATINOPOULOS, P. 2016. Valuing the benefits of an urban park project: A contingent valuation study in Thessaloniki, Greece. *Land Use Policy,* 55**,** 130-141.

LAUGHLAND, A. S., MUSSER, W. N., SHORTLE, J. S. & MUSSER, L. M. 1996. Construct Validity of Averting Cost Measures of Environmental Benefits. *Land Economics,* 72**,** 100.

LEE, A. C. K., JORDAN, H. C. & HORSLEY, J. 2015. Value of urban green spaces in promoting healthy living and wellbeing: prospects for planning. *Risk Management and Healthcare Policy,* 8**,** 131-137.

LEE, A. C. K. & MAHESWARAN, R. 2011. The health benefits of urban green spaces: A review of the evidence. *Journal of Public Health,* 33**,** 212-222.

LEIGH, R. & KUHNEL, I. 2001. Hailstorm loss modelling and risk assessment in the Sydney region, Australia. *Natural Hazards,* 24**,** 171-185.

LEKUTHAI, A. & VONGVISESSOMJAI, S. 2001. Intangible flood damage quantification. *Water Resources Management,* 15**,** 343-362.

LIEBMAN, M., PAYNE, N. & MOLTENO, J. 2015. On-site versus off-site-the business case for stormwater treatment in infill areas in Blacktown. *9th International Water Sensitive Urban Design (WSUD 2015)***,** 275.

LINDSEY, G. & KNAAP, G. 1999. Willingness to pay for urban greenway projects. *Journal of the American Planning Association,* 65**,** 297-313.

LIU, C. & LI, X. 2012. Carbon storage and sequestration by urban forests in Shenyang, China. *Urban Forestry & Urban Greening,* 11**,** 121-128.

LO, A. Y. & JIM, C. Y. 2010. Willingness of residents to pay and motives for conservation of urban green spaces in the compact city of Hong Kong. *Urban Forestry & Urban Greening,* 9**,** 113-120.

LOUVIERE, J. J., HENSHER, D. A., SWAIT, J. D. & ADAMOWICZ, W. 2000. *Stated choice methods: Analysis and application,* Cambridge, Cambridge University Press.

LOVELACE, J. & STRAUSER, C. 1998. Protecting society from flood damage: A case study from the 1993 upper Mississippi River flood.

LUTTIK, J. 2000. The value of trees, water and open space as reflected by house prices in the Netherlands. *Landscape and Urban Planning,* 48**,** 161-167.

MACDONALD, D. H., ARDESHIRI, A., ROSE, J. M., RUSSELL, B. D. & CONNELL, S. D. 2015. Valuing coastal water quality: Adelaide, South Australia metropolitan area. *Marine Policy,* 52**,** 116-124.

MACDONALD, D. H., BARNES, M., BENNETT, J., MORRISON, M. & YOUNG, M. D. 2005. Using a choice modelling approach for customer service standards in urban water. *Journal of the American Water Resources Association,* 41**,** 719-728.

MACDONALD, D. H., MORRISON, M. D. & BARNES, M. B. 2010. Willingness to pay and willingness to accept compensation for changes in urban water customer service standards. *Water Resources Management,* 24**,** 3145.

MAHMOUDI, P., HATTON MACDONALD, D., CROSSMAN, N. D., SUMMERS, D. M. & VAN DER HOEK, J. 2013. Space matters: the importance of amenity in planning metropolitan growth. *Australian Journal of Agricultural and Resource Economics,* 57**,** 38-59.

MAKEPEACE, D. K., SMITH, D. W. & STANLEY, S. J. 1995. Urban stormwater quality: Summary of contaminant data. *Critical Reviews in Environmental Science and Technology,* 25**,** 93-139.

MANSFIELD, C., PATTANAYAK, S. K., MCDOW, W., MCDONALD, R. & HALPIN, P. 2005. Shades of Green: Measuring the value of urban forests in the housing market. *Journal of Forest Economics,* 11**,** 177-199.

MARELLA, G. & RAGA, R. 2014. Use of the Contingent Valuation Method in the assessment of a landfill mining project. *Waste Management,* 34**,** 1199-1205.

MARSH, D. & BASKARAN, R. 2009. Valuation of water quality improvements in the Karapiro catchment: a choice modelling approach. *Australian Agricultural and Resource Economics Society 53rd Annual Conference.* Cairns, Australia.

MARTÍNEZ-PAZ, J. & PERNI, A. 2011. Environmental cost of groundwater: A contingent valuation approach. *International Journal of Environmental Research,* 5**,** 603-612.

MCCLELLAND, G. H., SCHULZE, W. D., LAZO, J. K., WALDMAN, D. M., DOYLE, J. K., ELLIOTT, S. R., IRWIN, J. R. & CARLIN, A. 1992. For measuring non-use values: A contingent valuation study of groundwater cleanup, draft report to USEPA. *Centre for Economic Analysis, University of Colorado.* Boulder, Colorado.

MCPHERSON, E. G. & SIMPSON, J. R. 2003. Potential energy savings in buildings by an urban tree planting programme in California. *Urban Forestry & Urban Greening,* 2**,** 73-86.

MEKALA, G. D., JONES, R. N. & MACDONALD, D. H. 2015. Valuing the benefits of creek rehabilitation: Building a business case for public investments in urban green infrastructure. *Environmental Management,* 55**,** 1354-1365.

MELBOURNE WATER 2013. Water sensitive urban design Life cycle costing data. Melbourne: Melbourne Water, State Government Victoria.

MELL, I. C., HENNEBERRY, J., HEHL-LANGE, S. & KESKIN, B. 2013. Promoting urban greening: Valuing the development of green infrastructure investments in the urban core of Manchester, UK. *Urban Forestry & Urban Greening,* 12**,** 296-306.

MENNEN, S. 2017. *The most cost-effective ways to maintain public open space with less water in Perth.* Masters, University of Western Australia.

MEYER, V., SCHWARZE, R., BECKER, N., MARKANTONIS, V., VAN DEN BERGH, J. C., BOUWER, L. M., BUBECK, P., CIAVOLA, P., GENOVESE, E. & GREEN, C. 2014. Assessing the Costs of Natural Hazards–State of the Art and the Way Forward. *Hydrometeorological Hazards: Interfacing Science and Policy***,** 253-290.

MIDDEL, A., HÄB, K., BRAZEL, A. J., MARTIN, C. A. & GUHATHAKURTA, S. 2014. Impact of urban form and design on mid-afternoon microclimate in Phoenix Local Climate Zones. *Landscape and Urban Planning,* 122**,** 16-28.

MIDDELMANN-FERNANDES, M. H. 2010. Flood damage estimation beyond stage–damage functions: an Australian example. *Journal of Flood Risk Management,* 3**,** 88-96.

MILLER, N., POGUE, D., GOUGH, Q. & DAVIS, S. 2009. Green buildings and productivity. *Journal of Sustainable Real Estate,* 1**,** 65-89.

MILLWARD, A. A. & SABIR, S. 2011. Benefits of a forested urban park: What is the value of Allan Gardens to the city of Toronto, Canada? *Landscape and Urban Planning,* 100**,** 177-188.

MINNESOTA STORMWATER STEERING COMMITTEE 2005. *The Minnesota stormwater manual,* USA, Minnesota Pollution Control Agency.

MOLINOS-SENANTE, M., HERNÁNDEZ-SANCHO, F. & SALA-GARRIDO, R. 2010. Economic feasibility study for wastewater treatment: A cost–benefit analysis. *Science of The Total Environment,* 408**,** 4396-4402.

MOLINOS-SENANTE, M., HERNÁNDEZ-SANCHO, F. & SALA-GARRIDO, R. 2011. Cost–benefit analysis of water-reuse projects for environmental purposes: A case study for Spanish wastewater treatment plants. *Journal of Environmental Management,* 92**,** 3091-3097.

MOLINOS-SENANTE, M. & SALA-GARRIDO, R. 2017. How much should customers be compensated for interruptions in the drinking water supply? *Science of The Total Environment,* 586**,** 642-649.

MORANCHO, A. B. 2003. A hedonic valuation of urban green areas. *Landscape and Urban Planning,* 66**,** 35-41.

MORRISON, M., BENNETT, J., BLAMEY, R. & LOUVIERE, J. 2002. Choice modeling and tests of benefit transfer. *American Journal of Agricultural Economics,* 84**,** 161-170.

MULLEN, J. D., CALHOUN, K. C. & COLSON, G. J. 2017. Preferences for policy attributes and willingness to pay for water quality improvements under uncertainty. *Water Resources Research,* 53**,** 2627-2642.

NAKAYAMA, T. & HASHIMOTO, S. 2011. Analysis of the ability of water resources to reduce the urban heat island in the Tokyo megalopolis. *Environmental Pollution,* 159**,** 2164-2173.

NAM, P. & SON, T. 2005. Households Demand for Improved Water Services in Ho Chi Minh City: A Comparison of Contingent Valuation and Choice Modelling Estimates. *The Economy and Environment Program for South East Asia (EEPSEA), Singapore.*

NAUGES, C., STRAND, J. & WALKER, I. 2009. The value of water connections in Central American cities: a revealed preference study. *Environment and Development Economics,* 14**,** 349-370.

NETUSIL, N. R., LEVIN, Z., SHANDAS, V. & HART, T. 2014. Valuing green infrastructure in Portland, Oregon. *Landscape and Urban Planning,* 124**,** 14-21.

NHDES 2012. New Hampshire Stormwater Manual. Concord, N.H, USA: New Hampshire Department of Environmental Services.

NICHOLLS, S. & CROMPTON, J. L. 2005. The impact of greenways on property values: Evidence from Austin, Texas. *Journal of Leisure Research,* 37**,** 321.

NIJNIK, M., ZAHVOYSKA, L., NIJNIK, A. & ODE, A. 2009. Public evaluation of landscape content and change: Several examples from Europe. *Land Use Policy,* 26**,** 77-86.

NORTH, J. H. & GRIFFIN, C. C. 1993. Water source as a housing characteristic: Hedonic property valuation and willingness to pay for water. *Water Resources Research,* 29**,** 1923-1929.

NORTON, B. A., COUTTS, A. M., LIVESLEY, S. J., HARRIS, R. J., HUNTER, A. M. & WILLIAMS, N. S. G. 2015. Planning for cooler cities: A framework to prioritise green infrastructure to mitigate high temperatures in urban landscapes. *Landscape and Urban Planning,* 134**,** 127-138.

NOWAK, D. J., APPLETON, N., ELLIS, A. & GREENFIELD, E. 2017. Residential building energy conservation and avoided power plant emissions by urban and community trees in the United States. *Urban Forestry & Urban Greening,* 21**,** 158-165.

NOWAK, D. J. & CRANE, D. E. 2002. Carbon storage and sequestration by urban trees in the USA. *Environmental Pollution,* 116**,** 381-389.

NOWAK, D. J., CRANE, D. E. & STEVENS, J. C. 2006. Air pollution removal by urban trees and shrubs in the United States. *Urban forestry & urban greening,* 4**,** 115-123.

NOWAK, D. J., HIRABAYASHI, S., BODINE, A. & GREENFIELD, E. 2014. Tree and forest effects on air quality and human health in the United States. *Environmental Pollution,* 193**,** 119-129.

NOWAK, D. J., HIRABAYASHI, S., BODINE, A. & HOEHN, R. 2013. Modeled PM2.5 removal by trees in ten U.S. cities and associated health effects. *Environmental Pollution,* 178**,** 395-402.

NUNES, P. A. L. D. & VAN DEN BERGH, J. C. J. M. 2001. Economic valuation of biodiversity: sense or nonsense? *Ecological Economics,* 39**,** 203-222.

PANDIT, R. & LABAND, D. N. 2010a. Energy savings from tree shade. *Ecological Economics,* 69**,** 1324-1329.

PANDIT, R. & LABAND, D. N. 2010b. A hedonic analysis of the impact of tree shade on summertime residential energy consumption. *Arboriculture and Urban Forestry,* 36**,** 73-80.

PANDIT, R., POLYAKOV, M. & SADLER, R. 2014. Valuing public and private urban tree canopy cover. *Australian Journal of Agricultural and Resource Economics,* 58**,** 453-470.

PANDIT, R., POLYAKOV, M., TAPSUWAN, S. & MORAN, T. 2013. The effect of street trees on property value in Perth, Western Australia. *Landscape and Urban Planning,* 110**,** 134-142.

PARKER, D. J. (ed.) 2000. *Floods,* London, UK: Routledge.

PARSONS, G. R., HELM, E. C. & BONDELID, T. 2003. Measuring the economic benefits of water quality improvements to recreational users in six northeastern states: an application of the random utility maximization model. *Working paper.* University of Delaware.

PARSONS, G. R. & KEALY, M. J. 1992. Randomly Drawn Opportunity Sets in a Random Utility Model of Lake Recreation. *Land Economics,* 68**,** 93.

PATTANAYAK, S. K., YANG, J.-C., WHITTINGTON, D. & BAL KUMAR, K. C. 2005. Coping with unreliable public water supplies: Averting expenditures by households in Kathmandu, Nepal. *Water Resources Research,* 41.

PAYNE, E. G. I., HATT, B. E., DELETIC, A., DOBBIE, M. F., MCCARTHY, D. T. & CHANDRASENA, G. I. 2015. Adoption Guidelines for Stormwater Biofiltration Systems. *Melbourne, Australia: Cooperative Research Centre for Water Sensitive Cities*.

PENG, M. & OLESON, K. L. 2017. Beach Recreationalists' Willingness to Pay and Economic Implications of Coastal Water Quality Problems in Hawaii. *Ecological Economics,* 136**,** 41-52.

PEPPER, C., MCCANN, L. & BURTON, M. 2005. Valuation study of urban bushland at Hartfield Park, Forrestfield, Western Australia. *Ecological Management & Restoration,* 6**,** 190-196.

PERRATON, S. C., BLACKWELL, B. D., FISCHER, A., GASTON, T. F. & MEYERS, G. D. 2015. Systemic barriers to wastewater reuse in Australia: some jurisdictional examples. *Australasian Journal of Environmental Management,* 22**,** 355-372.

PETOUSI, I., FOUNTOULAKIS, M. S., STENTIFORD, E. I. & MANIOS, T. 2015. Farmers' Experience, Concerns and Perspectives in Using Reclaimed Water for Irrigation in a Semi-Arid Region of Crete, Greece. *Irrigation and Drainage,* 64**,** 647-654.

PETRONE, K. C. 2010. Catchment export of carbon, nitrogen, and phosphorus across an agro-urban land use gradient, Swan-Canning River system, southwestern Australia. *Journal of Geophysical Research,* 115.

PFLÜGER, Y., RACKHAM, A. & LARNED, S. 2010. The aesthetic value of river flows: An assessment of flow preferences for large and small rivers. *Landscape and Urban Planning,* 95**,** 68-78.

PHANEUF, D. J. & SMITH, V. K. 2005. Chapter 15 Recreation Demand Models. *In:* MÄLER, K. G. & VINCENT, J. R. (eds.) *Handbook of Environmental Economics.* North Holland, Amsterdam: Elsevier.

PLANT, L., RAMBALDI, A. & SIPE, N. 2017. Evaluating Revealed Preferences for Street Tree Cover Targets: A Business Case for Collaborative Investment in Leafier Streetscapes in Brisbane, Australia. *Ecological Economics,* 134**,** 238-249.

POE, G. L. & BISHOP, R. C. 1992. Measuring the benefits of groundwater protection from agricultural contamination: results from a two-stage contingent valuation study.

POE, G. L. & BISHOP, R. C. 2001. Information and the Valuation of Nitrates in Ground Water, Portage County, Wisconsin. *In:* BERGSTROM, J., BOYLE, K. & POE, G. (eds.) *The Economic Value of Water Quality.* Northampton, MA: Edward Elgar.

POLYAKOV, M., FOGARTY, J., ZHANG, F., PANDIT, R. & PANNELL, D. J. 2016. The value of restoring urban drains to living streams. *Water Resources and Economics*.

POLYAKOV, M., WHITE, B. & ZHANG, F. 2017. Cost-effective Strategies to Reduce Nitrogen and Phosphorus Emissions in an Urban River Catchment. Melbourne, Australia (Draft). Melbourne: Cooperative Research Centre for Water Sensitive Cities.

POLYZOU, E., JONES, N., EVANGELINOS, K. I. & HALVADAKIS, C. P. 2011. Willingness to pay for drinking water quality improvement and the influence of social capital. *The Journal of Socio-Economics,* 40**,** 74-80.

POMEROY, J., GREEN, M. & FITZGIBBON, J. 1983. Evaluation of urban riverscape aesthetics in the Canadian prairies. *Journal of environmental management,* 17**,** 263-276.

POUDYAL, N. C., HODGES, D. G. & MERRETT, C. D. 2009a. A hedonic analysis of the demand for and benefits of urban recreation parks. *Land Use Policy,* 26**,** 975-983.

POUDYAL, N. C., HODGES, D. G., TONN, B. & CHO, S. H. 2009b. Valuing diversity and spatial pattern of open space plots in urban neighborhoods. *Forest Policy and Economics,* 11**,** 194-201.

POWELL, J. R., ALLEE, D. J. & MCCLINTOCK, C. 1994. Groundwater protection benefits and local community planning: Impact of contingent valuation information. *American Journal of Agricultural Economics,* 76**,** 1068.

PRODUCTIVITY COMMISSION 2011. Australia's urban water sector. *Inquiry report*.

RANJAN, R. 2014. Groundwater management through collective participation: why some institutions succeed and others fail? *Annals of Public and Cooperative Economics,* 85**,** 427-452.

RESSURREIÇÃO, A., GIBBONS, J., DENTINHO, T. P., KAISER, M., SANTOS, R. S. & EDWARDS-JONES, G. 2011. Economic valuation of species loss in the open sea. *Ecological Economics,* 70**,** 729-739.

RIES, R., BILEC, M. M., GOKHAN, N. M. & NEEDY, K. L. 2006. The Economic Benefits of Green Buildings: A Comprehensive Case Study. *The Engineering Economist,* 51**,** 259-295.

RINAUDO, J.-D. & AULONG, S. 2014. Defining groundwater remediation objectives with cost-benefit analysis: does it work? *Water resources management,* 28**,** 261-278.

ROLDÁN, E., GÓMEZ, M., PINO, M. R. & DÍAZ, J. 2015. The impact of extremely high temperatures on mortality and mortality cost. *International journal of environmental health research,* 25**,** 277-287.

ROLFE, J. & PRAYAGA, P. 2007. Estimating values for recreational fishing at freshwater dams in Queensland. *The Australian Journal of Agricultural and Resource Economics,* 51**,** 157-174.

ROLLINS, K., ZACHARIAH, O., FREHS, J. & TATE, D. 1997. Resource valuation and public policy: Consumers' willingness to pay for improving water servicing infrastructure. *Canadian Water Resources Journal,* 22**,** 185-195.

ROSADO, M. A., CUNHA-E-SA, M. A., DUCLA-SOARES, M. M. & NUNES, L. C. 2006. Combining averting behavior and contingent valuation data: an application to drinking water treatment in Brazil. *Environment and Development Economics,* 11**,** 729-746.

ROSENBERGER, R. S. & LOOMIS, J. B. 2000. Using meta-analysis for benefit transfer: In-sample convergent validity tests of an outdoor recreation database. *Water Resources Research,* 36**,** 1097-1107.

ROSSETTI, J. O. E. 2013. Valuation of Australia's green infrastructure: Hedonic pricing model using the enhanced vegetation index.

ROY, A. H., WENGER, S. J., FLETCHER, T. D., WALSH, C. J., LADSON, A. R., SHUSTER, W. D., THURSTON, H. W. & BROWN, R. R. 2008. Impediments and Solutions to Sustainable, Watershed-Scale Urban Stormwater Management: Lessons from Australia and the United States. *Environmental Management,* 42**,** 344-359.

RUPÉREZ-MORENO, C., PÉREZ-SÁNCHEZ, J., SENENT-APARICIO, J. & DEL PILAR FLORES-ASENJO, M. 2015. The economic value of conjoint local management in water resources: Results from a contingent valuation in the Boquerón aquifer (Albacete, SE Spain). *Science of The Total Environment,* 532**,** 255-264.

SALATA, F., GOLASI, I., PETITTI, D., DE LIETO VOLLARO, E., COPPI, M. & DE LIETO VOLLARO, A. 2017. Relating microclimate, human thermal comfort and health during heat waves: An analysis of heat island mitigation strategies through a case study in an urban outdoor environment. *Sustainable Cities and Society,* 30**,** 79-96.

SALDÍAS, C., SPEELMAN, S., DRECHSEL, P. & VAN HUYLENBROECK, G. 2017. A livelihood in a risky environment: Farmers' preferences for irrigation with wastewater in Hyderabad, India. *Ambio,* 46**,** 347-360.

SALE, M., HOSKING, S. & DU PREEZ, M. 2009. Application of the contingent valuation method to estimate a recreational value for the freshwater inflows into the Kowie and the Kromme Estuaries. *Water SA,* 35.

SANDER, H., POLASKY, S. & HAIGHT, R. G. 2010. The value of urban tree cover: A hedonic property price model in Ramsey and Dakota Counties, Minnesota, USA. *Ecological Economics,* 69**,** 1646-1656.

SANDER, H. A. & HAIGHT, R. G. 2012. Estimating the economic value of cultural ecosystem services in an urbanizing area using hedonic pricing. *Journal of Environmental Management,* 113**,** 194-205.

SANDER, H. A. & POLASKY, S. 2009. The value of views and open space: Estimates from a hedonic pricing model for Ramsey County, Minnesota, USA. *Land Use Policy,* 26**,** 837-845.

SANDHU, H. & WRATTEN, S. 2013. Ecosystem Services in Farmland and Cities. *Ecosystem Services in Agricultural and Urban Landscapes***,** 1-15.

SAPHORES, J. D. & LI, W. 2012. Estimating the value of urban green areas: A hedonic pricing analysis of the single family housing market in Los Angeles, CA. *Landscape and Urban Planning,* 104**,** 373-387.

SCARM 2000. Floodplain management in Australia: best practice principles and guidelines. Australia: Standing Committee on Agriculture and Resource Management, CSIRO Publishing.

SEIFERT, I., THIEKEN, A. H., MERZ, M., BORST, D. & WERNER, U. 2009. Estimation of industrial and commercial asset values for hazard risk assessment. *Natural Hazards,* 52**,** 453-479.

SETO, K. C., GÜNERALP, B. & HUTYRA, L. R. 2012. Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *Proceedings of the National Academy of Sciences,* 109**,** 16083-16088.

SHRESTHA, R., ROSENBERGER, R. & LOOMIS, J. 2007. Benefit transfer using meta-analysis in recreation economic valuation. *In:* NAVRUD, S. & READY, R. (eds.) *Environmental value transfer: Issues and methods.* Springer.

SHULTZ, S. D. & LINDSAY, B. E. 1990. The willingness to pay for groundwater protection. *Water Resources Research,* 26**,** 1869-1875.

SINGH, A., SYAL, M., GRADY, S. C. & KORKMAZ, S. 2010. Effects of green buildings on employee health and productivity. *American journal of public health,* 100**,** 1665-1668.

SMITH, D. 1994. Flood damage estimation- A review of urban stage-damage curves and loss functions. *Water S. A.,* 20**,** 231-238.

SMYTH, R., MISHRA, V. & QIAN, X. 2008. The environment and well-being in urban China. *Ecological Economics,* 68**,** 547-555.

SNOWBALL, J. D., WILLIS, K. G. & JEURISSEN, C. 2008. Willingness to pay for water service improvements in middle-income urban households in South Africa: A stated choice analysis. *South African Journal of Economics,* 76**,** 705-720.

SOARES, A. L., REGO, F. C., MCPHERSON, E. G., SIMPSON, J. R., PEPER, P. J. & XIAO, Q. 2011. Benefits and costs of street trees in Lisbon, Portugal. *Urban Forestry & Urban Greening,* 10**,** 69-78.

SOLLER, J. A. 2006. Use of microbial risk assessment to inform the national estimate of acute gastrointestinal illness attributable to microbes in drinking water. *Journal of Water and Health,* 04**,** 165.

STENGER, A. & WILLINGER, M. 1998. Preservation value for groundwater quality in a large aquifer: a contingent-valuation study of the Alsatian aquifer. *Journal of Environmental Management,* 53**,** 177-193.

STEVENS, T. H., BARRETT, C. & WILLIS, C. E. 1997. Conjoint analysis of groundwater protection programs. *Agricultural and Resource Economics Review,* 26**,** 229-236.

STREINER, C. F. & LOOMIS, J. B. 1995. Estimating the benefits of urban stream restoration using the hedonic price method. *Rivers,* 5**,** 267-278.

STROHBACH, M. W. & HAASE, D. 2012. Above-ground carbon storage by urban trees in Leipzig, Germany: Analysis of patterns in a European city. *Landscape and Urban Planning,* 104**,** 95-104.

SUGIYAMA, T., LESLIE, E., GILES-CORTI, B. & OWEN, N. 2008. Associations of neighbourhood greenness with physical and mental health: do walking, social coherence and local social interaction explain the relationships? *Journal of Epidemiology and Community Health,* 62**,** e9-e9.

SUN, H., BERGSTROM, J. C. & DORFMAN, J. H. 1992. Estimating the benefits of groundwater contamination control. *Southern Journal of Agricultural Economics,* 24**,** 63-71.

SUNSHINE COAST REGIONAL COUNCIL 2009. Flooding and Stormwater Management. *Discussion Paper.* Australia.

SUSCA, T., GAFFIN, S. R. & DELL'OSSO, G. R. 2011. Positive effects of vegetation: Urban heat island and green roofs. *Environmental Pollution,* 159**,** 2119-2126.

SUTHERLAND, R. J. & WALSH, R. G. 1985. Effect of Distance on the Preservation Value of Water Quality. *Land Economics,* 61**,** 281.

TALLIS, M., TAYLOR, G., SINNETT, D. & FREER-SMITH, P. 2011. Estimating the removal of atmospheric particulate pollution by the urban tree canopy of London, under current and future environments. *Landscape and Urban Planning,* 103**,** 129-138.

TAM, V. W. Y., TAM, L. & ZENG, S. X. 2010. Cost effectiveness and tradeoff on the use of rainwater tank: An empirical study in Australian residential decision-making. *Resources, Conservation and Recycling,* 54**,** 178-186.

TAMANG, P. & JANA, S. K. 2017. Willingness to Pay for Improved Water Services: A Case of Darjeeling, India. *Asian Journal of Water, Environment and Pollution,* 14**,** 51-59.

TAPSUWAN, S., BRENNAN, D., INGRAM, G. & BURTON, M. Household willingness to pay to avoid drought water restrictions: A case study of Perth. The 36 th Australian Conference of Economists, 2007 Hobart.

TAPSUWAN, S., BURTON, M., MANKAD, A., TUCKER, D. & GREENHILL, M. 2014. Adapting to less water: household willingness to pay for decentralised water systems in urban Australia. *Water resources management,* 28**,** 1111-1125.

TARFASA, S. & BROUWER, R. 2013. Estimation of the public benefits of urban water supply improvements in Ethiopia: a choice experiment. *Applied Economics,* 45**,** 1099-1108.

TENTES, G. & DAMIGOS, D. 2012. The Lost Value of Groundwater: The Case of Asopos River Basin in Central Greece. *Water Resources Management,* 26**,** 147-164.

TERVONEN, J., ALASAARELA, E. & SVENTO, R. 1994. Household water quality and consumer welfare: an application to the city of Oulu. *Aqua Fennica,* 24**,** 83-92.

TESFAYE, A. & BROUWER, R. 2015. Exploring the scope for transboundary collaboration in the Blue Nile river basin: downstream willingness to pay for upstream land use changes to improve irrigation water supply. *Environment and Development Economics,* 21**,** 180-204.

THE STATE OF QUEENSLAND 2002. Guidance on the assessment of tangible flood damages. Brisbane: Department of Natural Resources and Mines, The State of Queensland.

THE STATE OF VICTORIA DEPARTMENT OF ENVIRONMENT, L., WATER AND PLANNING, 2015 Valuing Victoria’s Parks : Accounting for ecosystems and valuing their benefits.

THOMPSON, K. D., STEDINGER, J. R. & HEATH, D. C. 1997. Evaluation and presentation of dam failure and flood risks. *Journal of Water Resources Planning and Management,* 123**,** 216-227.

THUNBERG, E. & SHABMAN, L. 1991. Determinants of landowner's willingness to pay for flood hazard reduction. *Journal of the American Water Resources Association,* 27**,** 657-665.

TORELL, L. A., LIBBIN, J. D. & MILLER, M. D. 1990. The market value of water in the Ogallala aquifer. *Land Economics,* 66**,** 163.

TRAN, Y. L., SIRY, J. P., BOWKER, J. M. & POUDYAL, N. C. 2017. Atlanta households’ willingness to increase urban forests to mitigate climate change. *Urban Forestry & Urban Greening,* 22**,** 84-92.

TU, G. Y., ABILDTRUP, J. & GARCIA, S. 2016. Preferences for urban green spaces and peri-urban forests: An analysis of stated residential choices. *Landscape and Urban Planning,* 148**,** 120-131.

TUDOR, D. T. & WILLIAMS, A. T. 2008. Important Aspects of Beach Pollution to Managers: Wales and the Bristol Channel, UK. *Journal of Coastal Research,* 243**,** 735-745.

TUSSUPOVA, K., BERNDTSSON, R., BRAMRYD, T. & BEISENOVA, R. 2015. Investigating Willingness to Pay to Improve Water Supply Services: Application of Contingent Valuation Method. *Water,* 7**,** 3024-3039.

TZIAKIS, I., PACHIADAKIS, I., MORAITAKIS, M., XIDEAS, K., THEOLOGIS, G. & TSAGARAKIS, K. P. 2009. Valuing benefits from wastewater treatment and reuse using contingent valuation methodology. *Desalination,* 237**,** 117-125.

TZOULAS, K., KORPELA, K., VENN, S., YLI-PELKONEN, V., KAŹMIERCZAK, A., NIEMELA, J. & JAMES, P. 2007. Promoting ecosystem and human health in urban areas using green infrastructure: a literature review. *Landscape and urban planning,* 81**,** 167-178.

UM, M.-J., KWAK, S.-J. & KIM, T.-Y. 2002. Estimating willingness to pay for improved drinking water quality using averting behavior method with perception measure. *Environmental and Resource Economics,* 21**,** 285-300.

VAN HOUTVEN, G., HONEYCUTT, A., GILMAN, B. & MCCALL, N. 2008. Costs of illness among older adults: An analysis of six major health conditions with significant environmental risk factors. RTI International.

VAN HOUTVEN, G., POWERS, J. & PATTANAYAK, S. K. 2007. Valuing water quality improvements in the United States using meta-analysis: Is the glass half-full or half-empty for national policy analysis? *Resource and Energy Economics,* 29**,** 206-228.

VAN HOUTVEN, G. L., PATTANAYAK, S. K., USMANI, F. & YANG, J.-C. 2017. What are Households Willing to Pay for Improved Water Access? Results from a Meta-Analysis. *Ecological Economics,* 136**,** 126-135.

VANDERMEULEN, V., VERSPECHT, A., VERMEIRE, B., VAN HUYLENBROECK, G. & GELLYNCK, X. 2011. The use of economic valuation to create public support for green infrastructure investments in urban areas. *Landscape and Urban Planning,* 103**,** 198-206.

VÁSQUEZ, W. F. & ESPAILLAT, R. 2016. Willingness to pay for reliable supplies of safe drinking water in Guatemala: A referendum contingent valuation study. *Urban Water Journal,* 13**,** 284-292.

VÁSQUEZ, W. F., MOZUMDER, P., HERNÁNDEZ-ARCE, J. & BERRENS, R. P. 2009. Willingness to pay for safe drinking water: Evidence from Parral, Mexico. *Journal of Environmental Management,* 90**,** 3391-3400.

VENTURA, S. J. & KIM, K. 1993. Modeling urban nonpoint source pollution with a geographic information system. *Journal of the American Water Resources Association,* 29**,** 189-198.

VERLICCHI, P., AL AUKIDY, M., GALLETTI, A., ZAMBELLO, E., ZANNI, G. & MASOTTI, L. 2012. A project of reuse of reclaimed wastewater in the Po Valley, Italy: Polishing sequence and cost benefit analysis. *Journal of Hydrology,* 432-433**,** 127-136.

VESELY, É.-T. 2007. Green for green: The perceived value of a quantitative change in the urban tree estate of New Zealand. *Ecological Economics,* 63**,** 605-615.

VIANA, D., GORNIK, K., LIN, C.-C., MCDONALD, G., NG, N. S., QUIGLEY, C. & POTOSKI, M. 2017. Recreational boaters value biodiversity: The case of the California Channel Islands National Marine Sanctuary. *Marine Policy,* 81**,** 91-97.

VO, D. T. & HUYNH, K. V. 2017. Estimating residents' willingness to pay for groundwater protection in the Vietnamese Mekong Delta. *Applied Water Science,* 7**,** 421.

VOTSIS, A. 2017. Planning for green infrastructure: The spatial effects of parks, forests, and fields on Helsinki's apartment prices. *Ecological Economics,* 132**,** 279-289.

WANG, H., XIE, J. & LI, H. 2010. Water pricing with household surveys: A study of acceptability and willingness to pay in Chongqing, China. *China Economic Review,* 21**,** 136-149.

WARD, F. & BEAL, D. 2000. Valuing nature with travel cost models. Edward Elgar Publishing.

WELSCH, H. 2002. Preferences over prosperity and pollution: Environmental valuation based on happiness surveys. *Kyklos,* 55**,** 473-494.

WHITE, L. 2008. *Sea the value: quantifying the value of marine life to divers.* Masters, Duke University.

WHITE, P. A., SHARP, B. M. & KERR, G. N. 2001. Economic valuation of the Waimea Plains groundwater system. *Journal of Hydrology(New Zealand),* 40**,** 59-76.

WHITTINGTON, D. 2002. Household demand for improved piped water services: evidence from Kathmandu, Nepal. *Water Policy,* 4**,** 531-556.

WHITTINGTON, D., BRISCOE, J., MU, X. & BARRON, W. 1990. Estimating the Willingness to Pay for Water Services in Developing Countries: A Case Study of the Use of Contingent Valuation Surveys in Southern Haiti. *Economic Development and Cultural Change,* 38**,** 293-311.

WILLIS, K. G., SCARPA, R. & ACUTT, M. 2005. Assessing water company customer preferences and willingness to pay for service improvements: A stated choice analysis. *Water Resources Research,* 41.

WOLDEMARIAM, G., SEYOUM, A. & KETEMA, M. 2016. Residents' willingness to pay for improved liquid waste treatment in urban Ethiopia: results of choice experiment in Addis Ababa. *Journal of Environmental Planning and Management,* 59**,** 163-181.

WONG, N. H. & YU, C. 2005. Study of green areas and urban heat island in a tropical city. *Habitat International,* 29**,** 547-558.

WONG, T. H. F., ALLEN, R., BROWN, R. R., DELETIĆ, A., GANGADHARAN, L., GERNJAK, W., JAKOB, C., JOHNSTONE, P., REEDER, M. & TAPPER, N. 2013. blueprint2013–stormwater management in a water sensitive city. *Melbourne, Australia: Cooperative Research Centre for Water Sensitive Cities, ISBN.* Melbourne, Australia.

WRIGHT, A. P. & HUDSON, D. Applying a Voluntary Incentive Mechanism to the Problem of Groundwater Conservation: An Experimental Approach. 2013 Annual Meeting, February 2-5, 2013, Orlando, Florida, 2013. Southern Agricultural Economics Association.

XIAO, Y., LI, Z. G. & WEBSTER, C. 2016. Estimating the mediating effect of privately-supplied green space on the relationship between urban public green space and property value: Evidence from Shanghai, China. *Land Use Policy,* 54**,** 439-447.

YAMASHITA, S. 2002. Perception and evaluation of water in landscape: use of Photo-Projective Method to compare child and adult residents' perceptions of a Japanese river environment. *Landscape and Urban Planning,* 62**,** 3-17.

YANG, J., MCBRIDE, J., ZHOU, J. & SUN, Z. 2005. The urban forest in Beijing and its role in air pollution reduction. *Urban Forestry & Urban Greening,* 3**,** 65-78.

YANG, J., YU, Q. & GONG, P. 2008. Quantifying air pollution removal by green roofs in Chicago. *Atmospheric Environment,* 42**,** 7266-7273.

YU, C. & HIEN, W. N. 2006. Thermal benefits of city parks. *Energy and Buildings,* 38**,** 105-120.

ZHAI, G. & IKEDA, S. 2006. Flood risk acceptability and economic value of evacuation. *Risk analysis,* 26**,** 683-694.

ZHANG, F., IFTEKHAR, M. S., BURTON, M., KININMONTH, I. & FOGARTY, J. 2015a. Understanding social preferences to avoid land use conflicts in wastewater treatment plant odor buffer zones. *2nd Water Sensitive Cities Conference.* Brisbane, Australia.

ZHANG, F., POLYAKOV, M., FOGARTY, J. & PANNELL, D. J. 2015b. The capitalized value of rainwater tanks in the property market of Perth, Australia. *Journal of Hydrology,* 522**,** 317-325.

ZHANG, Y., VAN DIJK, T., TANG, J. & BERG, A. 2015c. Green Space Attachment and Health: A Comparative Study in Two Urban Neighborhoods. *International Journal of Environmental Research and Public Health,* 12**,** 14342.

ZMIROU, D., PENA, L., LEDRANS, M. & LETERTRE, A. 2003. Risks associated with the microbiological quality of bodies of fresh and marine water used for recreational purposes: summary estimates based on published epidemiological studies. *Archives of Environmental Health: An International Journal,* 58**,** 703-711.

ŽUVELA-ALOISE, M., KOCH, R., BUCHHOLZ, S. & FRÜH, B. 2016. Modelling the potential of green and blue infrastructure to reduce urban heat load in the city of Vienna. *Climatic Change,* 135**,** 425-438.

CRC for
Water Sensitive Cities

Cooperative Research Centre for Water Sensitive Cities

Level 1, 8 Scenic Boulevard
Monash University
Clayton VIC 3800

info@crcwsc.org.au

www.watersensitivecities.org.au