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Ecosystem Service Impacts of Urban Water Supply and Demand Management

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Abstract Utilities face the challenge of enhancing long-term water security while minimising undesirable economic, social and environmental impacts of supply and demand management options. This paper provides an example of how the ecosystem services concept can be used to enumerate and organise broad impacts of water supply options. A case study of Adelaide, South Australia, is used to examine costs and benefits associated with different sources of water and source-water mix scenarios. Ecosystem service impacts are estimated using estimates from the literature. Seven water supply and demand management options are considered for Adelaide: 1) the River Murray, 2) Mt. Lofty Ranges catchments, 3) wastewater reuse, 4) desalination, 5) stormwater harvesting, 6) groundwater and 7) water conservation. The largest costs are associated with sourcing water from conservation measures such as water restrictions on outdoor watering estimated at $1.87/kL. Salinity damage costs associated with residential uses are estimated at up to $1.54/kL. Salinity damage costs of wastewater reuse were estimated at $1.16/kL. The largest benefit is coastal amenity services associated with stormwater harvesting and treatment estimated at $1.03/kL. Results show that there is a trade-off between financial costs and ecosystem services impacts with source-water mix scenarios with the highest ecosystem services cost having the lowest financial O&M cost and vice versa. This highlights the importance of taking ecosystem services into account when evaluating water supply options.

Keywords Water security · Water supply options · Environmental impacts · Externalities · Demand management

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1 Introduction

Worldwide governments face on-going challenges of how to enhance water security for increasingly urbanised populations. For cities that suffer economic or physical water scarcity, investments in multiple sources of water are likely to be required. This suggests a complex draw on water sources with the potential to impact not only the immediate urban landscape but large natural areas beyond metropolitan boundaries (McDonald et al. 2014). Rather than evaluate prospective water sources in isolation, a comprehensive evaluation of urban water supply sources should consider alternative plausible scenarios consisting of various mixes of existing and prospective water sources. A mechanism for doing this is to employ an ecosystem service (ES) framework in the context of new sources augmenting an existing water supply system. The contribution of this paper is to demonstrate how to evaluate a broad set of financial and environmental impacts for realistic scenarios for an urban water supply system. We use a case study area, Adelaide, South Australia where there are seven potential sources of water of varying quality and impacts.

A transdisciplinary literature has emerged which highlights the magnitude of beneficial services provided by ecosystems (Costanza et al. 2014). According to this literature, decision-making and resource utilisation needs to take into account the impact on ecosystems and the stock and flow of beneficial services for the wellbeing of people on the planet (Daily et al. 2009). Operationalising the ecosystem service concept requires the assembly of economic values linked to relevant changes in available ecosystem services (ES) associated with various water supply options and source-water mixes. This involves reviewing published studies and undertaking new work to provide probable ranges of values. In this paper, the ecosystem service (ES) approach is based on a wider social-ecological view of cities (Garcia et al. 2016) consistent with integrated water resource management - the latter being a familiar concept to water resource managers (Mitchell et al. 2007).

The objective of the analysis in this paper is to identify and, where feasible, quantify impacts by estimating the marginal costs and benefits associated with seven water supply and demand management options (hereafter called *options*) and three different mixes of source-water for metropolitan Adelaide in South Australia. Methods for adapting the ES framework to the scale of a metropolitan area and applying it across the urban water cycle are outlined. We assess broad ES impacts as well as financial operations and maintenance (O&M) costs of water supply and unintended financial costs of water supply and demand management imposed on water consumers including the cost of coping with water restrictions and salinity damage costs (hereafter called *financial costs*). The seven options under consideration include water from: 1) River Murray, 2) Mt. Lofty Ranges catchments, 3) wastewater reuse, 4) desalination, 5) stormwater harvesting, 6) groundwater and 7) demand management. We considered three near-optimal water mixes for Metropolitan Adelaide identified in Maheepala et al. (2014) that: 1) makes greatest use of the conventional existing surface water sources to minimise financial O&M cost of supply; 2) makes greatest use of treated wastewater and harvested stormwater for non-potable uses so as to reduce the adverse environmental impact of discharging these sources to the environment; and 3) represents a compromise solution that has reasonable cost and reasonable levels of discharge of stormwater and wastewater to the environment (respectively).

This paper is organised with a discussion of the theoretical context, methods used to estimate the ES values a case study context and a description of the methodology for enumerating impacts of multiple water sources. Estimates of the costs, benefits and lessons learned from application of the method are presented. Finally, key conclusions drawn from the analysis are summarised.

2 Theoretical Context

Potential impacts of the seven water supply and demand management options for Adelaide were organised using the ES typology. Building on the Millennium Ecosystem Assessment, Kumar (2010) categorised ES broadly as provisioning, cultural, regulating, supporting, and habitat services. *Provisioning services* are products such as water, food and fibre; *cultural services* include benefits such as recreational, educational and spiritual services; *regulating services* provide the means to moderate or regulate air, water and climate quality; *supporting services* maintain the condition of the ecology such as nutrient and soil cycling services; *habitat support services* constitute ES that support plant and animal habitats as well as genetic diversity.

Underlying the ES concept is the anthropocentric view that people derive value from environmental assets in ecological systems and this can be measured through the contribution to human well-being. We are intentional in our use of the word "value" as opposed to "price" as value is a broader concept which envelops direct and indirect use, options and insurance for the future, existence and bequest values. Methods have been established for estimating the value of ES (Bateman et al., 2011) as changes in net social welfare resulting from changes in the stock of natural assets and the flow of services from either the production or consumption side (termed, *producer* and *consumer surplus* respectively). In principle, these methods try to elicit the economic value of losses or gains from a decrease or an increase in the quality of ES defined not as the market price for ES, but rather as the sum of: 1) the amount beneficiaries would be willing to pay for the additional ES or quality-gain in ES, or to avoid a decrease in the quality of ES of known magnitude (consumer surplus); and 2) the amount producers would be willing to accept for the additional ES or for the better-quality ES (producer surplus) (Bateman et al. 2011). On the production side, use values can be estimated with market prices excluding taxes, subsidies and/or market power being exerted by firms or labour. Where multiple inputs are used, a production function approach can be used e.g. value of irrigation water in agricultural production.

3 Case Study Area and Policy Context

Adelaide, the capital of the state of South Australia, is a semi-arid coastal city with 1.2 million people and variable rainfall averaging about 549 mm per year. Water supply headworks for the Adelaide metropolitan region consist of an integrated system of ten reservoirs in the Mt. Lofty Ranges that collect and store water from local catchments (Fig. 1). The total storage capacity of the reservoirs is 200 GL. This is supplemented by water pumped via three major pipelines from the River Murray some 90 km east of the city.

The annual water consumption of Adelaide is in the range 150 to 180 GL. Up until the year 2000, the percentage of supply to Adelaide from the River Murray varied between 10% and 90% with an average of 35% (ATSE, 2012). During the Millennium Drought of 2000–2009, the available supply from both the River Murray and Mt. Lofty Ranges catchments was severely limited. As a consequence, the government of South Australia imposed increasingly severe

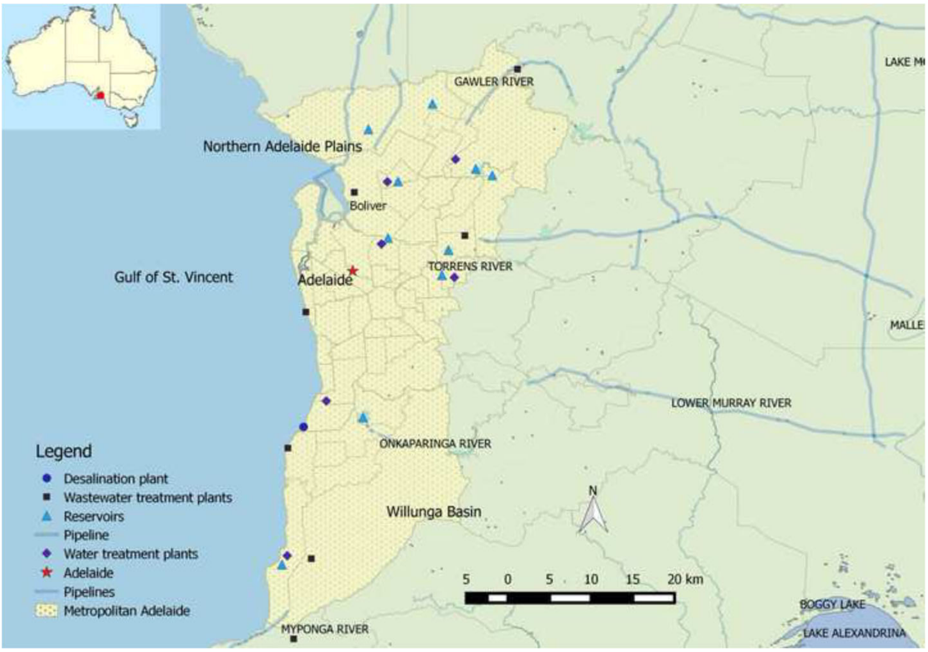

Fig. 1 Map of the case study area

demand management measures including restrictions on outdoor water use, particularly the watering of gardens. As the drought deepened, the Commonwealth and State governments invested in a 100GL new desalination plant. Local councils commissioned stormwater harvesting projects to provide water for industry, irrigate parks and reduce pollutant discharge to coastal waters. In addition, the three main wastewater treatment plants (WWTP) are capable of providing treated wastewater for non-potable use including irrigated agriculture in peri-urban areas, watering private and public gardens and flushing toilets. Finally, up to 6% of water consumption in metropolitan Adelaide is sourced from groundwater resources (SAWater 2012).

Excess stormwater, wastewater and brine from the desalination plant, are discharged to Gulf St Vincent (Fig. 1). Stormwater and wastewater have contributed towards a decline in the quality of Adelaide's coastal waters over a number of decades (Russell et al. 2009). The Gulf waters are characterised by a rich variety of marine flora and fauna with seagrass meadows that provide habitats for rare species with important biodiversity values. A range of marine and coastal provisioning, regulating, habitat and cultural ES may be impacted by these discharges, including fish production, amenity and recreation (Luisetti et al. 2011).

4 Methods

Our methodology involves seven distinct steps: 1) enumerating ES impacts of water supply and demand management; 2) defining the water supply scenario under the current delivery strategy for water; 3) defining the changes under various alternative mixes of water sources; 4) reviewing of peer-reviewed studies to obtain ranges of values for parameters used to estimate broad ecosystem service impacts of water supply options and source-water mixes; 5) calculating ecosystem service costs and financial costs common to all water options and source water mixes; 6) comparing ES impacts of water supply and demand management options; and 7) sensitivity analysis of key variable parameters using Monte Carlo simulation.

4.1 Enumerating Potential ES Impacts

The six water supply and one demand management option for Adelaide were identified with state government stakeholders. ES associated with the urban water cycle for each water source were identified. Details for stormwater are provided in previous work (Tables 2 and 3 of Kandulu et al. 2014) and details of ES identified for other water sources can be found in online supplementary material Appendices A, B and C.

4.2 Defining the Base Case Water Supply Scenario

The baseline water consumption in the Adelaide metropolitan area is defined as the pattern observed in 2012/13 (Table 1). This year was chosen for pragmatic reasons relating to the desalination plant coming online and several stormwater interception projects being operational. The mix of the water sources used in 2012/13 is summarised in Table 2 and volumes are given in the “Baseline” column in Table 2.

Baseline water consumption was categorised as either residential or non-residential and further as either for potable or non-potable uses.

4.3 Defining Source Water mix Scenarios

Three possible mixes of water sources are summarised in Table 2 (based on an options study described in Maheepala et al. 2014). Mix 1 makes greatest use of the conventional sources (Mt Lofty Ranges and River Murray) and less use of reclaimed wastewater and harvested stormwater than the baseline. The aim of this mix is to minimize the financial O&M cost of supply by making greater use of existing conventional surface water sources including Mt. Lofty Ranges and River Murray. Mix 2 represents a compromise between minimising cost and minimising the discharge of untreated storm- and wastewater to marine ecosystems in Gulf St Vincent. It uses more water from the Mt. Lofty Ranges, harvested stormwater and recycled wastewater and less water from the River Murray than the Baseline. Mix 2 also minimises use of water

Table 1 Annual water consumption for Metropolitan Adelaide in 2012/13

Component	Volume (ML)
Residential Potable	45,510
Residential Non-Potable	70,577
Total Residential	116,086
Non-Residential Potable	25,137
Non-Residential Non-Potable	29,232
Total Non-Residential	54,369
Total Potable	70,647
Total Non-Potable	99,809
TOTAL	170,456

Table 2 Percentage supply in 2012/13 from the various water supply sources including Mt. Lofty Ranges (ML), River Murray (RM), desalination (Desal), stormwater (SW) and wastewater (WW) for Metropolitan Adelaide

Component	Percentage Supply from each Source			
	Baseline	Mix 1	Mix 2	Mix 3
ML: Potable	14.6	24.2	30.7	36.1
ML: Non-Potable	14.8	30.5	21.6	12.2
RM: Potable	16.9	17.3	11.0	5.5
RM: Non-Potable	17.1	21.8	7.8	1.8
Desal: Potable	10.2	0.2	0.0	0.1
Desal: Non-Potable	10.2	0.2	0.0	0.1
SW: Non-Potable	5.9	2.7	7.0	11.7
WW: Non-Potable	10.3	3.2	21.9	32.5

from desalination, the most energy intensive source. The aim of Mix 3 is to minimise the discharge of stormwater and wastewater to Gulf St Vincent. It uses harvested stormwater and recycled wastewater to the greatest extent of all of the mixes and makes the least use of the River Murray. It is aimed at achieving the maximum use of alternative sustainable water sources. The use of desalinated water is small for all of the cases except for the baseline as the desalination plant will largely operate at low levels except in drought or in the face of strong population growth.

4.4 Systematic Review

A programme evaluation approach was utilised (Bateman et al. 2011) which facilitates a systematic approach to marginal changes in ES. Capital costs associated with existing infrastructure were not included in this analysis as these were deemed to be sunk costs. Additional capital costs may be relevant in a context where existing infrastructure is close to capacity. A full set of 30 ES impacts were considered. Twenty-one of these ES impacts are quantified and listed in Table 3. Three ES impacts were assessed qualitatively. In six cases, neither qualitative nor quantitative assessment of ES impacts could be carried out due to insufficient information being available.

A systematic review of the literature was carried out using Google, Google Scholar and Web of Science using the commonly used terms for each water source, relevant ES and the case study location. Then the search was widened to combinations of water sources and ES to identify ranges of values for key ES and estimates of relevant ES cost and benefit values. Figure 2 summarises the broad range of ES values that were identified in this paper with the ES estimates classified into either consumer or producer surplus values (to facilitate benefit-cost analysis).

The objective of the literature search was to locate quantitative estimates from diverse sources in particular published literature and consulting reports. Empirical studies focusing on the Adelaide metropolitan area or other Australian cities were preferred so as to avoid potential differences in cultural settings and world-views. Point estimate transfers, relevant formulas or estimated functions were extracted from studies and formed the basis of the range of values given in Table 3.

Where multiple estimates and functions were identified, a range of values is presented along with sources. To demonstrate the approach used, salinity and

Table 3 Parameter descriptions, transfers, value ranges and sources

Parameter or ES value	Type of Transfer	Unit	Value Range	Source(s)
1 Mt. Lofty Ranges catchments				
Provisioning food and fibre	Adapted from Qureshi et al. (2010)	A$/kL	0.06–0.15	(Qureshi et al. 2010)
Salinity levels, *T*		mg/L	80–900	(Kandulu et al. 2014)
Climate regulation		kWh/kL	0.3	(ATSE, 2012)
2 River Murray				
Provisioning food and fibre	Formula from same growing region	A$/kL	0.08–0.15	(Qureshi et al. 2010)
Salinity levels, *T*		mg/L	279–828	(Page et al. 2013)
Climate regulation		kWh/kL	1.9	(ATSE, 2012)
Habitat services	Modified to new ecology	A$/kL	0.03	(Hatton MacDonald et al. 2011; MDBA 2011; Pollino et al. 2011)
Recreational amenity		A$/kL	0.06	(Boyd and Banzhaf 2007)
3 Wastewater reuse				
Provisioning food and fibre	Adapted from Qureshi et al. (2010)	A$/kL	0.13–0.15	(Qureshi et al. 2010)
Salinity levels, *T*		mg/L	1098–1200	(Kandulu et al. 2014; Page et al. 2013)
Climate regulation		kWh/kL	0.69–1.06	(Kenway et al. 2008; Marchi et al. 2014)
Water quality regulation		A$/kL	0.07	(Dillon et al. 2009; Hall 2012; SAEPA 2011)
4 Desalination				
Salinity levels, *T*		mg/L	160	(Kandulu et al. 2014)
Climate regulation		kWh/kL	5–5.43**	(DWLBC 2005; Marchi et al. 2014)
5 Stormwater harvesting				
Coastal amenity services	Transfer of a formula	$/kL	1.03	(Kandulu et al. 2014)
Salinity levels, *T*		mg/L	125–240	(Kandulu et al. 2014)
Estuarine habitat services		A$/kL	0.00–0.05	(Kandulu et al. 2014)
Cultural amenity services		A$/kL	0.02	(Kandulu et al. 2014)
6 Groundwater				
Provisioning food and fibre	Adapted from Qureshi et al. (2010)	A$/kL	0.13–0.15	(Qureshi et al. 2010)
Climate regulation		kWh/kL	1.2	(SF 2008)
7 Conservation				
Provisioning food and fibre	Formula for time costs	A$/kL	0.59–1.87	(Brennan et al. 2007; Chong et al. 2009; Kandulu et al. 2014; Grafton and Ward 2008; Marchi et al. 2014)

* If not otherwise specified, a GHG emission factor equal to 0.79 $kgCO_{2\text{-}e}/kWh$ (DCCEE, 2012) and a GHG unit cost equal to $24.15*10^{-3}$ $A\$/kgCO_{2\text{-}e}$ (CEF, 2011; CER, 2015) have been used

**Includes the embedded energy of chemicals and membranes

***Further detail in electronic Appendix C

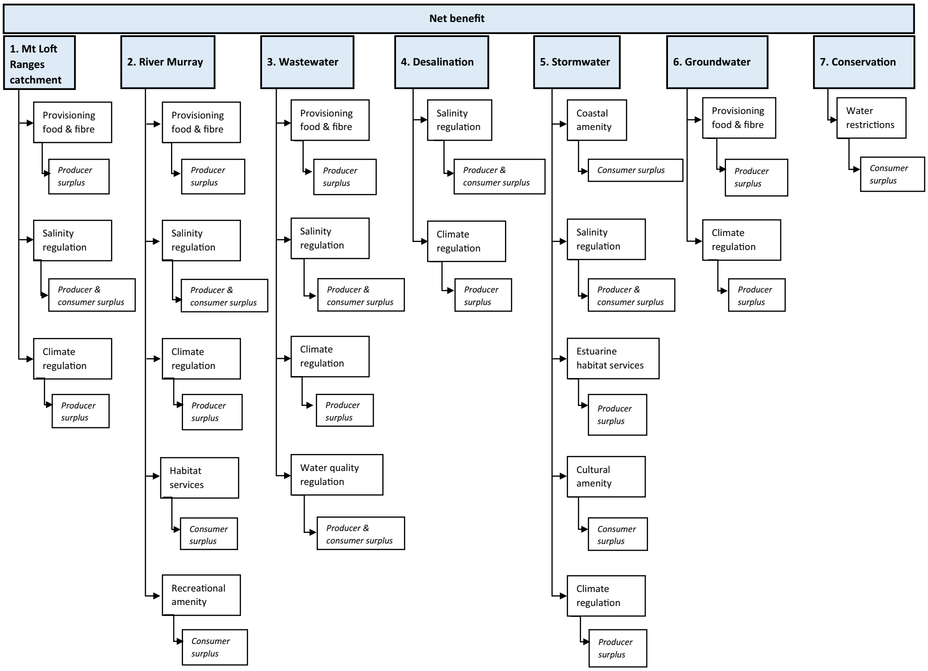

Fig. 2 Organisational structure for techniques used to quantify ES values of water supply and demand options for Adelaide

greenhouse gas calculations, common across options, are provided. We refer the reader to supplementary online support material for details on how other ecosystem service impacts were estimated.

4.5 Calculating Common Ecosystem Service Impacts

Salinity and GHG emission costs feature for all water supply options and source water mixes. In what follows, we describe how these costs were estimated in detail.

4.5.1 Salinity Costs

Salinity in treated mains water from each of the six water supply sources (Fig. 2) can damage residential and commercial hot water systems, water filters, pipes and plumbing fixtures. This can impose a cost on water users through increased replacement costs (ACG 2004). The loss in consumer surplus for residential supply was estimated as:

$$Household\ \ Cost\ \left(\frac{\$}{\frac{household}{yr}}\right) = 0.2458 \cdot T + 135 \tag{1}$$

where T is the salinity of water supplied measured in milligrams per litre.

For commercial users, producer surplus loss resulting from the cost of replacing damages from high salinity levels were estimated as:

$$Commercial\ \ Cost\left(\frac{\frac{\$}{kL}}{yr}\right)=0.00063{\cdot}T+0.35 \tag{2}$$

4.5.2 GHG Emission Costs

A large fraction of the greenhouse emissions (GHGs) originate in treatment and delivery of water and the embodied energy in the construction of infrastructure. As few additional capital works are planned, only operational GHGs in the consumption of energy were estimated using eq. (3)

$$GHG\ Cost\left(\frac{\$}{kL}\right)=Energy\left(\frac{kWh}{kL}\right){\cdot}Emission\ Factor\left(\frac{kgCO_{2-e}}{kWh}\right){\cdot}GHG\ Unit\ Cost\left(\frac{\$}{kgCO_{2-e}}\right) \tag{3}$$

4.6 Comparing ES Impacts of Water Supply and Demand Management Options

ES value estimates are in 2016 Australian dollars to adjusted using a CPI data (RBA 2016). The ES impacts of sourcing an additional kL of water from each of the seven options were estimated considering whether the water was going to be used for potable or non-potable uses. Further, the baseline scenario was compared with the estimated marginal costs (or benefits) over the volume of water that is re-allocated under each of the three source-water mixes. All costs have been computed as levelised costs per kL. The capital costs of all infrastructure existing in 2013 have been ignored as these are considered to be sunk costs.

4.7 Sensitivity Analysis

Monte Carlo simulation was used to generate frequency distributions of net cost estimates under the baseline scenario and under each of the three alternative source-water mix scenarios. The ES and financial costs were assumed to have uniform distributions for the purpose of this simulation. We also estimated the relative contribution of each uncertain parameter in Table 5 to allow for a range of estimates of net impacts under the three alternative source-water mix scenarios. Specifically, sensitivity analysis was conducted on cost, varying one uncertain parameter at a time (all other parameters being set at their respective median values).

5 Results

Working through the water sources, sourcing water from the River Murray in the water supply mix would: 1) reduce the amount of environmental water for recreational use downstream of the River Murray system ($0.06/kL), 2) reduce the flushing of the Ramsar sites of the Coorong

Table 4 A summary of salinity regulation and climate regulation costs (A$/kL) associated with water supply options for metropolitan Adelaide

Ecosystem service impact	Salinity regulation	Climate regulation (GHG emission
1. Mt. Lofty Ranges Catchments	1.37* 0.58^	0.01
2. River Murray	1.54* 0.65^	0.04
3. Wastewater reuse	1.16*	0.02
4. Desalination	1.15^ 0.48*	0.1
5. Stormwater harvesting	0.54* (with ASR*^) 0.46* (without ASR*^)	0.012
6. Groundwater		0.023

*Non-potable use

^ Potable use

*^ Aquifer storage and recharge

and Lower Lakes ($0.03/kL) which are below the Adelaide off-takes and 3) decrease the amount of agricultural production in the Murray-Darling Basin as a result of decreasing entitlements to irrigation water ($0.08/kL- $0.15/kL). Distribution and use of water sourced from the River Murray would have salinity impacts ($1.54/kL) and GHG associated with the electricity required ($0.04/kL). Similarly, increasing the use of water from the other surface water sources, such as the Mt. Lofty catchment, reduces water available for agriculture in the Adelaide Hills ($0.06/kL- $0.15/kL). The salinity cost to households from using Mt. Lofty Ranges catchment water will depend on whether it is used for potable ($0.58/kL) or non-potable uses ($1.37/kL). The recreational values associated with the Onkaparinga River downstream of storages was identified, but estimates of value were not readily transferable as existing studies in Australia focus on major rivers or wetlands. Estimates of ES values are summarised and reported in Tables 3 and 4.

Overall, the time costs imposed by water restrictions on outdoor water uses, requiring hand-watering rather than sprinklers, can incur a cost up to $1.87/kL. However this might be mitigated by the benefits of enhancing social cohesion through conservation (the latter benefits are not easily estimated). Salinity impacts were estimated at $1.54/kL (River Murray), $1.37/kL (Mt Lofty Ranges) and $1.15/kL (desalinated seawater) based on background salinity in these water sources (or remaining salinity in the case of desalinated water). The largest beneficial

Table 5 Costs for the various mixes of water supply for Metropolitan Adelaide

	Cost of Each Mix ($/kL)			
	Baseline	Mix 1	Mix 2	Mix 3
Cost of ES	1.04	1.11	1.06	1.00
Financial O&M Costs	0.67	0.39	0.70	0.88
Total Cost	1.71	1.50	1.75	1.88
Marginal Cost of ES	0.00	0.07	0.02	−0.04
Marginal Financial Cost	0.00	−0.28	0.02	0.21
Marginal Total Cost	0.00	−0.21	0.04	0.17

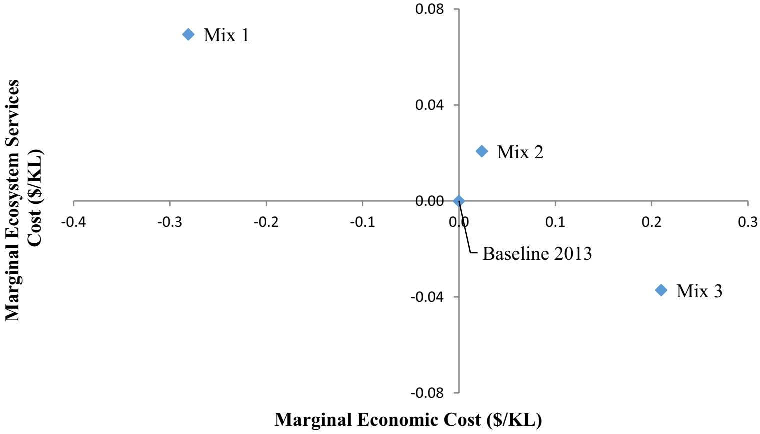

Fig. 3 Trade-offs between marginal ES cost and marginal economic cost for a number of mixes of water supply for Metropolitan Adelaide

impact is the coastal amenity associated with stormwater harvesting and treatment estimated at \$1.03/kL. The remaining ecosystem impacts were relatively small at less than \$0.20/kL.

The ES costs, financial O&M costs, total costs and the marginal values of each of the three alternative source-water mix scenarios (relative to the base case) are given in Table 5.

Table 5 demonstrates the trade-off between ES cost and the financial O&M cost of the various mixes with Mix 1 having the highest ES cost and the lowest financial O&M cost and Mix 3

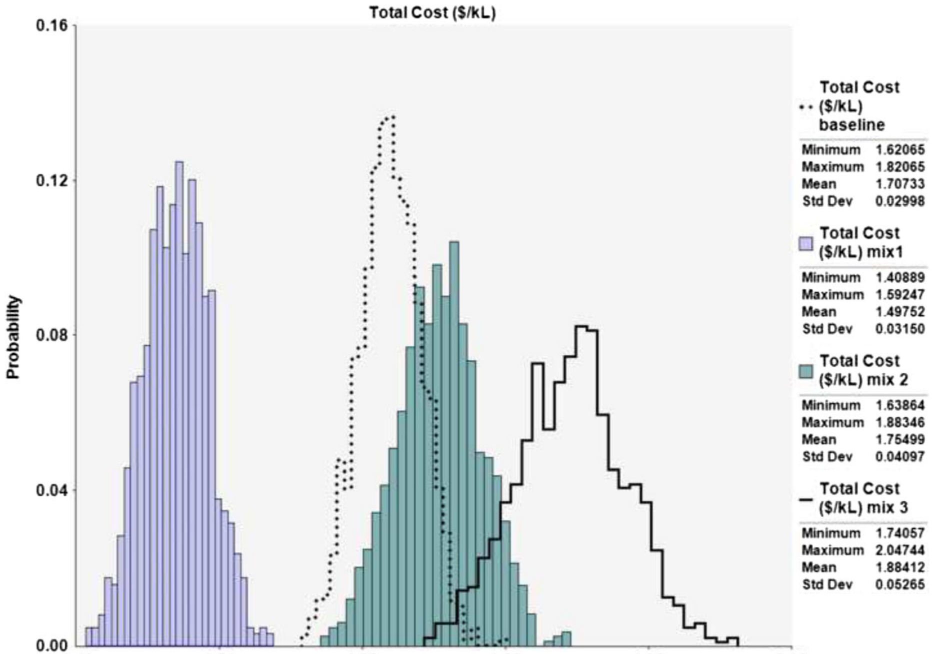

Fig. 4 Frequency distributions of simulated net costs under the baseline and three alternative source-water mix scenarios for Metropolitan Adelaide

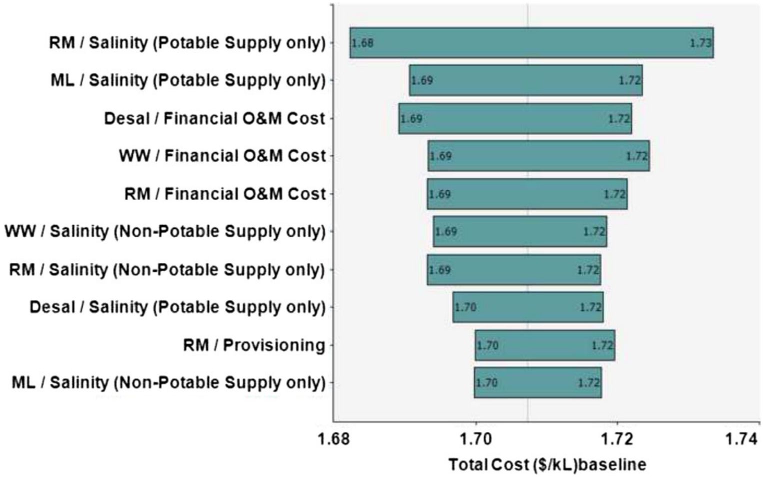

Fig. 5 Measuring the sensitivity of net return estimates for the baseline case to variability in uncertain parameters

having the lowest ES cost and the highest financial O&M cost. The trade-off between the marginal ES cost and the marginal financial O&M cost for the various mixes is shown in Fig. 3.

Figure 3 shows that the choice of the solution with the lowest financial O&M cost may be inferior from an ES perspective and vice versa. Shifting from the baseline source-water mix scenario to mix 3 would incur the highest incremental financial O&M costs, but also yield the highest ES benefit.

Descriptive statistics of the estimated net returns under the four scenarios are also provided in Fig. 4. Robust rankings based on net costs can be observed despite uncertainty in ES and financial cost estimates with the baseline being the least costly followed by mix 1, 2, and 3 respectively.

Figure 5 is a tornado chart showing results on measures of sensitivity of net return estimates for the baseline case to variability in each uncertain parameter holding other uncertain parameters at their median values.

Overall, net impacts were not sensitive to uncertainty except salinity damage cost estimates and financial O&M costs of water supply (e.g. River Murray, Mount Lofty Ranges and desalination). Recreational amenity and greenhouse gas emission costs of wastewater were also found to be an important contributor to variability in net impacts.

6 Discussion

Application of the ES typology to assess impacts of water supply and demand management options revealed some key limitations. For instance, assuming a linear relationship between ES values and volume of water may be a gross simplification. For example, if 5100 GL of environmental water is required to maintain the quality of downstream ecosystems in the River Murray, adding 100 GL onto low flows is unlikely to achieve even a proportional level of benefits.

Provisioning, habitat and regulation services in this context proved difficult to value in isolation as there is potential for joint production and double-counting. For example, stormwater harvesting could result in multiple environmental impacts such as improved water quality, reduced erosion and increased fish populations in receiving coastal waters. The value of fish production can be estimated using net returns from sales of extra fish production resulting coastal water quality improvements holding fishing effort constant. This profit is in part dependent on water quality. Aggregating the value of improved water quality and increased fish production resulting from stormwater harvesting would introduce double-counting errors (see Bateman et al. 2011).

The use of avoided costs and replacement costs to estimate impacts on ecosystem services has been criticized as not capturing the ‘use values’ of ecosystems (Bockstael et al. 2000; Rooney et al. 2015). Holland et al. (2010) contended that estimates of replacement costs do not fully capture economic welfare values (by definition based on actual choices of consumers and producers) of an ecosystem service and could under- or overestimate expected benefits. Further, there is methodological difficulties with the use of abatement costs to measure welfare as it requires the assumption that society would accept the costs of reducing pollution loads to avoid degradation.

As noted by Costanza et al. (2014), there is a degree of uncertainty in most estimated values associated with ES. The extent to which uncertainty matters depends on how estimated values will be used. If the objective is raising awareness, less precision may be required, whereas more precision may be required to devise optimal payments to landholders for improving ES. For the purposes of policy decisions, periodically investing in research will reduce the uncertainties around specific public policy decisions.

7 Conclusions

Urban water supply augmentation decisions that do not adequately consider environmental and social impacts could result in suboptimal outcomes. When ES costs were applied to a set of alternative mixes for supplying water to metropolitan Adelaide, it was demonstrated that there was a clear trade-off between the mix that gave the lowest financial O&M cost and the one that gave the lowest ES cost (and vice versa). The development of this trade-off highlights the challenges for decision-makers choosing among the various options. In the context of large infrastructure, there is a risk of adopting mal-adaptive investments and incurring irrecoverable high set-up costs. More information can aid in the optimal allocation of scarce public resources. A broad assessment of potential impacts will assist in supporting decision-making processes to incorporate environmental impacts beyond the traditional financial capital, operations and maintenance costs.

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