Using Life Cycle Assessment to Inform Total Water Cycle Management Planning - an Investigation for the Caboolture Catchment

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The Urban Water Security Research Alliance (UWSRA) is a $50 million partnership over five years between the Queensland Government, CSIRO's Water for a Healthy Country Flagship, Griffith University and The University of Queensland. The Alliance has been formed to address South East Queensland's emerging urban water issues with a focus on water security and recycling. The program will bring new research capacity to South East Queensland tailored to tackling existing and anticipated future issues to inform the implementation of the Water Strategy.

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Description: Caboolture region

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FOREWORD

Water is fundamental to our quality of life, to economic growth and to the environment. With its booming economy and growing population, Australia's South East Queensland (SEQ) region faces increasing pressure on its water resources. These pressures are compounded by the impact of climate variability and accelerating climate change.

The Urban Water Security Research Alliance, through targeted, multidisciplinary research initiatives, has been formed to address the region's emerging urban water issues.

As the largest regionally focused urban water research program in Australia, the Alliance is focused on water security and recycling, but will align research where appropriate with other water research programs such as those of other SEQ water agencies, CSIRO's Water for a Healthy Country National Research Flagship, Water Quality Research Australia, eWater CRC and the Water Services Association of Australia (WSAA).

The Alliance is a partnership between the Queensland Government, CSIRO's Water for a Healthy Country National Research Flagship, The University of Queensland and Griffith University. It brings new research capacity to SEQ, tailored to tackling existing and anticipated future risks, assumptions and uncertainties facing water supply strategy. It is a $50 million partnership over five years.

Alliance research is examining fundamental issues necessary to deliver the region's water needs, including:

- ensuring the reliability and safety of recycled water systems.
- advising on infrastructure and technology for the recycling of wastewater and stormwater.
- building scientific knowledge into the management of health and safety risks in the water supply system.
- increasing community confidence in the future of water supply.

This report is part of a series summarising the output from the Urban Water Security Research Alliance. All reports and additional information about the Alliance can be found at http://www.urbanwateralliance.org.au/about.html.

C. Davis

Chris Davis
Chair, Urban Water Security Research Alliance

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EXECUTIVE SUMMARY

The Urban Water Security Research Alliance (the Alliance) commissioned research into whether, and how, the Life Cycle Assessment (LCA) methodology could be used to inform the planning for urban water systems infrastructure in South East Queensland (SEQ).

This report is one of three produced to meet that goal. In this particular report, the focus is on the Total Water Cycle Management (TWCM) planning process, which is progressively being adopted by regional councils across the state. This report also provides a synthesis of findings across the other two reports produced by the LCA investigation, to the extent that they are relevant to the discussion on TWMCP. The role of the companion reports was to provide a greater level of detail in the analysis of conventional wastewater treatment systems (Lane *et al.* 2012b) and wastewater recycling (Lane *et al.* 2012a).

Improvements to the TWCM Plan Case Study Analysis

In the analysis undertaken for this report, LCA was applied to three scenarios developed for the recent Caboolture TWCM planning study of the Moreton Bay Regional Council (MBRC). Our scenario modelling used the best available empirical data, from a wide variety of sources. Using these 'improved' data led to different conclusions than if using the generic estimates for infrastructure operations that are more frequently adopted in SEQ planning studies.

Two key issues were identified for analysis of the non-traditional water supply systems (rainwater tanks and Class A+ water) that are increasingly being considered in SEQ. Firstly, there is insufficient empirical data available to support robust predictions of water usage by households that are serviced by these non-mains supplies. Secondly, in order to support meaningful greenhouse gas comparisons, decision makers should adopt the best available empirical data on electricity use, fugitive gas emissions, and chemical use. Recommended data sets for each of these are provided.

Appropriate definition of the system boundary is another critical element for comparative options analysis. The mains supply system must be included in the modelling of infrastructure scenarios, as the level of mains supply displacement could have a substantial effect on the life-cycle impacts associated with the choices that are made. However, the industry would require guidance on the most appropriate choices for modelling of the marginal mains supply source. In the absence of this, decision makers should consider the sensitivity of their options analysis to a range of different assumptions about the mains water source.

Extending the Environmental Scope of Decision Making

The analysis across all three project reports indicates that neither electricity use, nor greenhouse gas (GHG) accounting more generally, will be universally adequate as a proxy for the broader environmental tradeoffs associated with urban water planning decisions. Including a number of other LCA impact assessment metrics could give valuable insight into the broader implications of water system options under consideration. This would reduce the chances that decisions made now, but regarding infrastructure with long-term lifespans, could create an unexpected environmental burden that one day becomes an active concern for managers of the SEQ urban water system.

Depletion of the ozone layer is one particular issue that could come to the fore, because of the importance of nitrous oxide (N_2O) emissions to ozone layer chemistry, and the large N_2O fluxes that can be associated with urban wastewater systems. LCA also shows great potential to provide perspective on two emerging issues (phosphorus recovery; chemical toxicity from wastewater and biosolids contaminants) that are already recognised in the industry. Unfortunately, for both these issues, the available LCA models are not currently robust enough to provide meaningful input to urban water systems analysis.

There are a number of other LCA metrics that could be included in decision-making frameworks. This would require the collection of additional data beyond the basic water/nutrient/sediment balances that are the traditional focus. However the optimal data requirements for analysing this wider set of life-cycle impacts – power use, chemicals use, chemicals transport, sludge transport, fugitive gas emissions - are not much more than that needed for robust GHG analysis.

For LCA impact assessment metrics to be used in practice, decision makers would need some form of perspective on the relative importance of these 'new' environmental considerations. This can be achieved by benchmarking (or normalising) the results from a particular case study against some higher order reference data set. However, there is no one ideal approach, as the range of possible normalisation approaches would each provide a different set of useful insights.

Normalising the options analysis against higher-order (national and/or global) benchmarks will provide some understanding of broader social imperatives for impact prioritisation. Unfortunately, there are a number of important gaps in the datasets available for this normalisation approach, and users should beware of the potential for this to introduce significant bias into the process of options analysis. Nonetheless, normalisation of LCA results in this way can be useful for identifying those environmental issues that might be forced onto the radar of urban water system managers at some point in the future. Improved data and uncertainty assessment for these benchmark datasets could provide a strategic benefit for the urban water industry.

Benchmarking the TWCMP case study results against impact estimates for the whole of the regional urban water system provides a different set of perspectives that could also prove useful for the decision making process. An interim version of this benchmarking approach was developed through this project, and could be used as the basis for a more complete analysis of the entire SEQ urban water system.

1. INTRODUCTION

Project Context

This Urban Water Security Research Alliance project investigated a number of modelling approaches that could inform the Total Water Cycle Management (TWCM) planning process. The project team was tasked with reviewing the benefits of, and constraints on, using the Life Cycle Assessment (LCA) methodology for urban water systems planning.

This report is one of three developed to meet that goal. The reports are complementary, in that they each focus on a specific aspect of the LCA investigation. While they are structured so that each can be read as stand alone documents, there is much overlap in the case studies and methodology that were used. Detailed descriptions of the data and methodology are generally provided in only one of the three reports; hence a degree of cross-referencing is used.

The three reports are:

> *Using Life Cycle Assessment to inform Total Water Cycle Management Planning – an investigation for the Caboolture catchment* (this report) provides the parent study for the LCA investigation, and synthesises findings across all three reports. It also directly applies LCA to a specific TWCM plan case study for the Caboolture region in South East Queensland (SEQ), in order to illustrate the broader benefits and challenges in adopting such an approach.
>
> *Life Cycle perspectives on wastewater recycling* (Lane *et al.* 2012a) provides more detailed analysis of wastewater recycling options. While focussed on the Caboolture area, this report aims to inform a broader debate on the role of wastewater recycling in meeting the water supply needs for the rapidly growing population in SEQ. The wastewater report provides data that could be used in urban water systems planning studies, and highlights improvements to the LCA methodological framework that would enhance its capacity to inform the decision making process.
>
> *Application of Life Cycle Assessment to wastewater systems planning* (Lane *et al.* 2012b) provides a more detailed investigation into conventional urban sewage management systems, addressing a major gap in the TWCMP case study that is the focus of this report. The wastewater report provides data that could be used in wastewater systems planning studies, identifies a number of emerging issues that might become important to the wastewater industry over time, and highlights improvements to the LCA methodological framework that would enhance its capacity to inform the decision making process.

Total Water Cycle Management Planning

The Total Water Cycle Management Planning (TWCMP) process aims to integrate planning across the water supply, wastewater and stormwater components of the urban water system. The objective of TWCMP is to optimise the social and environmental benefits, and minimise the costs, across the system as a whole (WBD 2010). The benefits of TWCMP are particularly relevant in SEQ, given the enormous challenges involved in meeting current constraints on water use and pollution loads, at the same time as planning for a possible doubling in population by 2050 (QWC 2010).

The TWCMP concept provides a framework for consideration of environmental issues beyond the traditional hydrology/nutrient focus on local waterways. Greenhouse gas (GHG) emissions are now also being considered in much of the region's debate on urban water system planning. This reflects the growing influence of NGERS reporting requirements, the prospect of financial implications from a national carbon pricing scheme, and the realisation that the current investment trajectory in Australia is leading to urban water systems with a much higher GHG intensity than was previously the case (Kenway *et al.* 2008; Hall *et al.* 2009; Hall *et al.* 2011).

This increasing focus on GHG emissions does represent a broadening of the horizons for the urban water industry. However, there is very little consideration being given to what other environmental externalities might be of relevance to urban water systems decision making.

A Role for Life Cycle Assessment

A number of SEQ-based studies have previously used the LCA methodology to investigate different aspects of urban water systems. Originally, the focus was on wastewater treatment systems (de Haas *et al.* 2008; de Haas *et al.* 2009; Foley 2009). Subsequently, this was extended to consider an integrated, city-scale water supply and wastewater system at the Gold Coast (Lane *et al.* 2011). LCA has also been used to inform planning decisions on water system infrastructure in other large urban centres of Australia (Lundie *et al.* 2004; Grant *et al.* 2005; Peters *et al.* 2009; Grant *et al.* 2012).

This ongoing interest in the LCA approach reflects its capacity to take a long term view across a broad range of impacts, for comparisons of dissimilar and complex options. The application of LCA is underpinned by ISO standards (ISO 2006), and its strengths are threefold. Firstly, the LCA approach imposes rigour on the definition of system boundaries for options analysis. Secondly, it provides a framework for understanding key flows (such as water and nutrients) across the full life cycle of a system. Thirdly, it provides the most scientifically rigorous methodology available for analysis across such a broad spectrum of environmental sustainability issues. As a consequence, the use of LCA is growing rapidly across much of the world – both as a research tool, and also to inform policy making.

Research Objectives

The introduction of TWCMP represents a move in SEQ towards more rigorous and comprehensive analysis of urban water system options. The goal of the three reports is to identify if, and how, the LCA methodology could be used to further enhance the urban water planning process, and TWCMP in particular. To achieve this, the primary objectives for this report are to:

1. *Identify which LCA impact categories could be used to support TWCMP.*
2. *Use a real-life case study to explore how LCA could be used in the TWCMP process.*
3. *Identify the range of datasets that would be required to support analysis of these LCA impact categories.*

Chapter 2 provides conclusions on the choice of LCA impact categories, to satisfy objective 1. More detailed critique of specific LCA impact models is provided in the parallel reports on wastewater treatment (Lane *et al.* 2012b) and wastewater recycling (Lane *et al.* 2012a).

Chapters 3 and 4 address objectives 2 and 3 by applying the LCA methodology directly to a TWCMP case study.

This report also provides a summary of the project work with regard to an additional set of objectives:

4. *Make recommendations on energy, greenhouse gas and other data for use in TWCP studies.*
5. *Identify key data gaps that will constrain effective analysis of urban water systems in SEQ.*
6. *Use LCA to identify environmental issues that, while not currently considered a priority by the industry, may require a response from urban water planners at some point in the future.*

Chapters 3 and 4 highlight key datasets and data gaps of relevance to the chosen TWCMP case study, partially addressing objectives 4 and 5.

Chapter 6 provides a synthesis of the findings across this and the two other LCA-related reports produced by this project (see Lane *et al.* 2012a; Lane *et al.* 2012b). In doing so, this summary chapter provides additional comment on objectives 4 and 5, and provides a summary of findings of relevance to objective 6.

2. THE RELEVANCE OF LCIA MODELS FOR URBAN WATER SYSTEMS PLANNING

The review of available Life Cycle Impact Assessment (LCIA) models considered best practice guidelines from Australia (Grant *et al.* 2008; Bengtsson *et al.* 2010) and internationally (ECJRC 2011), recent publications in the scientific literature, and detailed discussions with international researchers involved in current projects on LCIA methodology development.

Further detail on the context for this review, and the LCIA approach to impact assessment, is provided in a previous UWSRA report (Lane *et al.* 2011). The two companion reports to this study (Lane *et al.* 2012a; 2012b) provide updated reviews for some of the impact categories under consideration.

The role of this chapter is to provide a summary of these investigations, also factoring in the learnings from the TWCMP analysis included in this report. The first section of this chapter provides a brief overview of the LCIA approach. The second section outlines a set of impact indicators and models that are most relevant and appropriate to the analysis of urban water systems in SEQ, as required to meet objective 1 for this report.

2.1. Life Cycle Impact Assessment

LCA has evolved to focus on three fundamental areas - the protection of natural environments, protection of human health, and maintenance of natural resources. Life Cycle Impact Assessment (LCIA) is the process of converting system inventory flows (emissions; resource extractions) into measures of potential impact in one or more of these areas.

The emission of a particular substance follows an often complex series of processes and pathways that may eventually result in some undesirable damage or degradation occurring. For example, the emission of carbon dioxide (CO_2) causes radiative forcing which results in climate change, and this is expected to eventually damage both natural ecosystems and human health.

The LCA methodology provides two options in terms of accounting for these impacts (Table 1). ***Endpoint*** indicators provide an estimate of ‘damage’ to physical elements that society wishes to protect, such as human health or the quality of natural environments, and are therefore closely aligned to the ultimate goals of environmental protection. ***Midpoint*** indicators are ‘measured’ at some intermediate point along the cause-effect chain between the stressor (e.g. the emission) and the damage (e.g. to human health). *Midpoint* indicators are typically intended to provide a compromise between the needs for relevance and low uncertainty.

Table 1: Summary of the Endpoint and Midpoint approaches to Life Cycle Impact Assessment.

	Midpoint	Endpoint
Approach	Measured at some point on the continuum (from inventory flow → endpoint damage) where the uncertainty is reduced.	Provide an estimate of the ultimate ‘damage’.
Application for greenhouse gas emissions	Global Warming Potential, which is a measure of the relative contribution of different gases (e.g. CH_4 vs. CO_2) to radiative forcing.	Separate assessment of climate change impacts on biodiversity and human health.
Benefits	Lower uncertainty. Midpoint indicators sometimes coincide with metrics used at policy and institutional level (e.g. *Global Warming Potential*).	Directly addresses the ultimate social concern. Disparate activities (e.g. greenhouse gas emissions vs. water use) can be directly compared in terms of the ultimate ‘damage’ (e.g. loss of biodiversity) that they cause.
Constraints	Increased number of indicators required, that can only be considered as measures of *potential* damage. Produces independent indicator results without a direct means for comparing the relative significance across the indicators.	Measures of endpoint damage normally involve much greater uncertainty than measures at intermediate points. The development of quantitative damage (endpoint) models for use in LCA lags considerably behind the development of midpoint indicator models.

Midpoint indicators were adopted for this study. This is the most commonly used approach in LCA analysis, and there has been substantial research effort put into developing midpoint-level impact assessment models. They can also be closely aligned to current management paradigms – e.g. *Global Warming Potential* is a midpoint indicator for climate change, which uses the same measurement concepts as conventional GHG accounting protocols promoted by the IPCC (IPCC 2006a).

Given the number, variation and global distribution of inventory flows associated with LCA studies, practical application requires that LCIA results are calculated using linear models (Equation 1). Water utilities will be familiar with this approach, as it is applied for greenhouse gas accounting. The same approach is used for all other midpoint impact categories used in LCA.

$$\text{Impact Potential} = \text{Impact Factor} \times \text{Inventory Flow}$$

$$\left(\textit{eg: } \underset{[kg\text{-}CO_2\ equiv]}{\textit{Global Warming Potential}} = \underset{[kg\text{-}CO_2 / kg\text{-}CH_4]}{GWP_{methane}} \times \underset{[kg\text{-}CH_4]}{\textit{emission of methane}}\right) \qquad \textit{Equation 1}$$

2.2. Summary of Recommendations

Table 2 provides a summary of the status and preferred choice of models for each of the LCA impact categories considered most relevant to analysis of urban water systems in SEQ. Reference is made to the other UWSRA reports that provide more detailed discussion on each individual impact category (Lane *et al.* 2012a; Lane *et al.* 2012b).

Table 2: Recommended LCIA models for use in urban water systems analysis

Impact Category	As proxy for…	Recommendation	Fidelity of the recommended model
Freshwater Extraction (FWE)[1]	ecosystem damage from disruptions to the hydrological cycle, and from discharge of nutrients/COD to waterways	WSI-weighted addition of freshwater extractions, excluding urban rainwater harvesting that occurs in the estuarine zone	Reasonable, although the resolution of the water stress index (WSI) modelling of Pfister et al (2009) needs to be critiqued by Australian eco-hydrological experts, and may need to be improved.
Eutrophication Potential (EP)[2]		aggregated metric for TN & TP emissions	Reasonable, if the majority of life-cycle emissions occur in SEQ coastal zones. Further review of nutrient status, speciation & dynamics is required. Airborne fate models are poor.
Ecotoxicity Potential (marine – METP; freshwater – FETP)[3]	ecosystem damage from discharge of organic and metal pollutants	exclude until improvements delivered	Current models are not able to adequately model the relative significance of metals and organic micropollutants
Ecotoxicity Potential (terrestrial – TETP)[2]			
Global Warming Potential (GWP)[1]	ecosystem & human health damage from changes to atmospheric concentrations of greenhouse gases and stratospheric ozone	use 100yr midpoint-impact factors from ReCiPe model	Best available
Ozone Depletion Potential (ODP)[2]		use impact factors provided in Lane et al (2012b)	Best available, although further research required to finalise impact factors for N_2O
Fossil Fuels Depletion (FD)[1]	availability of critical resources for use by future generations	use midpoint-impact factors from ReCiPe model	Best available
Minerals Depletion (MD)[2]		use midpoint-impact factors from ReCiPe model	Best available, but do not account for phosphate rock resources, therefore can't be used for analysis of phosphorus recovery
Human Toxicity Potential (HTP)[3]	human health damage by emissions of organic chemicals and metals (HTP); substances that cause smog (POFP), & substances that cause atmospheric particulate accumulation (PMFP)	use midpoint-impact factors from ReCiPe "I" version	Best available, but only addresses a limited portion of chemical exposure pathways related to urban systems
Photochemical Oxidant Formation Potential (POFP)		use midpoint-impact factors from ReCiPe	Best available, but model suitability requires further review
Particulate Matter Formation Potential (PMFP)		use midpoint-impact factors from ReCiPe	Best available, but model suitability requires further review

[1] see Lane et al(2011); [2] see Lane et al(2012b); [3] see Lane et al (2012a)

3. LCA FOR THE CABOOLTURE TWCMP

The LCA framework was applied to the Caboolture area scenarios developed as part of the Moreton Bay Regional Council (MBRC) TWCMP study (BMT-WBM 2012).

The LCA results are used to comment on certain differences between the scenarios, rather than make recommendations on the best choice of urban water infrastructure for the Caboolture area. The TWCMP decision criteria span many more issues than just environmental considerations, and therefore LCA results can only inform one part of the TWCMP decision making scope.

The objectives of this chapter are to:

- illustrate the types of insight that LCA could provide in the TWCMP context, and identify the data requirements for this to happen; and
- illustrate the implications of key data gaps to TWCMP scenario comparisons.

3.1. Case Study Overview

3.1.1. System Boundary

Scenarios

The TWCMP study (BMT-WBM 2012) identified three alternative scenarios that could be used to meet the planning requirements for the Caboolture River catchment area. Key elements of the Caboolture area urban system boundary are presented in Table 3.

These three scenarios (Table 4) represent an increasing level of sophistication and/or effort being invested in waterway and infrastructure management. They differ in their approach to wastewater reuse, stormwater treatment and reuse, rainwater tank implementation, and measures to reduce catchment sourced pollutant generation. However, the planning study assumed that the upgrade of the South Caboolture Sewage Treatment Plant (currently under way) is implemented and will accommodate any future population growth in its sewerage catchment. No variations on the level of this secondary wastewater treatment were considered.

Table 3: Caboolture area urban system boundary for the scenario analysis.

Expected population growth in the existing urban area[1]	42,700 (61%) increase, from current population of 69,500 to future population of 112,200
Mains water supply	The Caboolture Weir has a yield of 3.6 GL/y, which is already fully utilised. The area already imports ~ 11Gl/y of potable water from the SEQ Water Supply Grid.
Sewage treatment[2]	South Caboolture STP has been upgraded to service ~80,000 EP, with nutrient discharge limits of TN=2.5 mg/L, TP=0.3 mg/L.

1 This does not include the additional population (~60,000) to be accommodated in the Caboolture Identified Growth Area. Separate scenarios were developed for the greenfield development at the CIGA site, and this area was excluded from analysis in this report.
2 The Burpengary East STP processes sewage from the adjoining Burpengary Creek catchment, but discharges treated effluent into the lower reaches of the Caboolture River. This STP has been excluded from the analysis in this report.

Scenario 1 represents the minimum standard for water supply and stormwater management for new housing development in SEQ. The Queensland Development Code (QG 2008a) requires that all new housing must incorporate an alternative water supply option that reduces its demand on the mains water network. The most common way to meet this requirement is to install household rainwater tanks that supply toilet, laundry and external uses - this is the approach adopted for Scenario 1. Another planning constraint, as set out in the State Planning Policy 4/10 Healthy Waters (2010), is that stormwater controls for developments greater than 2500m^2 (or 5 lots) must be sufficient to remove 80% of the suspended solids in the stormwater, 60% of the phosphorus, and 45% of the nitrogen.

Scenario 1 assumes that sewage generation increases linearly with population growth, and that secondary treated effluent conditions are maintained at the design conditions for the upgrade currently occurring at the South Caboolture STP. The TWCMP does not formally consider STP biosolids management options, and so it was assumed that the biosolids are disposed of as landfill material, as was the practice at the time of this study.

Scenario 2 incorporates the elements of Scenario 1 with additional management actions. This includes a targeted approach to improve stormwater management on development sites, in order to reduce erosion and sediment losses. Also included is a scheme to use 8 ML/d of tertiary treated wastewater for agricultural irrigation, following an actual case study being investigated by Unitywater. No other changes were included for wastewater management, nor for urban water supplies. Scenario 2 also involves changes outside the urban area, primarily to reduce catchment erosion, through the implementation of agricultural best-management practices and rehabilitation of stream riparian zones.

Scenario 3 adopts all the elements of Scenario 2, except that it replaces the agricultural wastewater reuse system with a scheme for residential and non-residential urban reuse of Class A+ quality water[1]. Also included is a stormwater harvesting and treatment system that would reticulate water to a portion of the residential and commercial properties not serviced by the wastewater reuse network. It was assumed that the residential Class A+ water reuse (wastewater and stormwater) would be for toilet flushing and outdoor uses, meaning those households would also install rainwater tanks to provide water for laundry use. Scenario 3 also includes the implementation of more advanced stormwater treatment (WSUD) systems in both greenfield and existing urban areas. The catchment management initiatives of Scenario 2 were included without any change.

Table 4: Scenarios under consideration for the Caboolture catchment of the Moreton Bay Regional Council total water cycle management planning study.

Scenario	Catchment	Stormwater	Water Supply	Wastewater
1 *minimum requirements for new development*	--	TSS, TP, TN reduced by 80/60/45% (for new developments)	Rainwater Tanks to toilet, laundry & outside use (for new developments)	Linear growth (with population increase) in STP throughput and nutrient loads in treated wastewater
2* *easier advance beyond minimum requirements*	Best management practices (BMPs) for revegetation & farming in the Caboolture Rv. catchment	*as per Scenario 1* + Improved management of runoff from development sites	*as per Scenario 1*	*Sewage management as per scenario 1*
				Wastewater recycling to agriculture
3* *more ambitious objectives*	*as per Scenario 2*	*as per Scenario 2* + enhanced WSUD for existing & new areas	Rainwater Tanks to laundry use (for new developments)	*Sewage management as per scenario 1*
		Stormwater harvesting and reuse to urban centres		
			Class A+ wastewater reuse to urban areas	

* Scenario 2 builds on and extends the management options included in Scenario 1. Scenario 3 builds on and extends the management options included in Scenario 2.

Functional Unit and Life-Cycle Considerations

Where the scenarios are directly compared, the functional unit for the analysis is:

> "*the provision of water supply and stormwater management services to cater for a population increase of 42,700 in the Caboolture urban area*".

[1] The Queensland Water Quality Guidelines for Recycled Water Schemes (2008b) classifies recycled waters depending on the level of contaminants that are removed in the treatment process. Class A+ is the highest quality of recycled water for use in non-drinking purposes.

With regards to the urban water system infrastructure, the system boundary is varied for different sections of this chapter to determine what effect this might have on the outcomes from the case study options comparison. Section 3.3 explores the implications of making different choices on whether mains water supply is included, and the assumed source of that mains water. Section 3.4 expands the system boundary to assess the full urban water system, including the full degree of stormwater discharge and wastewater treatment associated with the full population increase.

The life-cycle modelling includes estimates for infrastructure construction and operations, but future infrastructure disposal in the end-of-life phase was excluded from the analysis. The end-of-life phase is typically found to have negligible effect on the LCA results for urban water systems (Gaterell *et al.* 2005; Vince *et al.* 2009). The analysis excludes the implications (e.g. residential water heating) of urban water use.

3.1.2. Foreground Data Collection

Where possible, the best available empirical data from local studies was used to estimate operational inventories for each of the scenario components. Where such data was not available, best estimates were informed by the latest science, otherwise taken from the information collected for a similar study on the Gold Coast area of South East Queensland (Lane *et al.* 2011). That Gold Coast study also informed our materials inventories for infrastructure construction.

Household and Non-Residential Water Use

Residential water use estimates were informed by the findings of end-use studies recently completed in SEQ (Beal *et al.* 2011; Stewart 2011). The basic per-capita end use assumptions are summarised in Table 5, and these match those used for the MBRC TWCMP study.

Differences that were introduced for the analysis in this report relate to the influence of non-mains supplies on household water usage. Further discussion on the modelling approach is provided in the companion report on wastewater recycling (Lane *et al.* 2012a).

For households connected to the Class A+ reuse schemes, it was assumed that the supply of treated wastewater or treated stormwater would not completely offset the demand for outdoor (e.g. gardening) mains water use. Conceptually, this recognises two possible influences: (a) that the introduction of an alternative supply might change the householder's perception on the need for water use minimisation; and (b) that householders might have a preference for using mains water for particular outdoor end uses. These possibilities were demonstrated in an early end use study on the Pimpama Class A+ system (Willis *et al.* 2009; Willis *et al.* 2011), however this issue has not been the focus of any subsequent empirical end use studies in SEQ.

For households connected only to rainwater tanks, we assumed that there would be no such effect, i.e., preferential use of mains water would not occur in this case. Treating rainwater supplies differently to Class A+ supplies is a somewhat arbitrary distinction, given the lack of empirical data available to support quantitative assumptions on this topic. For our analysis, this was based on the notion that householders would be less likely to hold a bias against the use of rainwater for outdoor purposes.

The main point of difference for the end-use modelling of households with rainwater tanks was that we assumed a much lower rainwater tank yield than was used for the MBRC TWCMP study. The source of this assumption is presented in the following section.

Our analysis did not consider the possibility that the presence of non-mains supplies (rainwater or Class A+ water) would increase the overall use of water for outdoor purposes. Such an outcome would in fact seem plausible, given the operators of A+ reuse schemes tend to encourage a strong uptake so as to reduce payback times on the capital infrastructure required. Other studies (Lane *et al.* 2011; Stewart 2011) have demonstrated that assumptions to this end could have a significant influence on urban water systems analysis.

We also excluded the possibility that household water leaks could increase the total flows to those end uses (toilets, laundry, outdoors) that are typically supplied by the non-mains systems considered in this study. Further review of the available empirical end use data would be required to determine how best to address this gap in quantitative planning and options analysis.

In general, end-use modelling for households with non-mains supplies is likely to be a complex exercise. The implications of our assumptions are summarised in Table 6. However, it would seem that there is insufficient empirical data available to support robust end-use analysis when such systems are under consideration.

Table 5: Basic residential end use assumptions for all scenarios.

end use	demand (L/p/d)	demand (L/hh/d)#
laundry	31	87
toilet	24	66
outdoor use	25	70
other indoor uses	75	209
leaks##	9	25
total	163	457

assuming an average of 2.8 people per household, as per the MBRC study

assuming that leaks do not affect toilet, laundry or outdoor usage[2]

Table 6: Water use supply and demand balances for (a) households with a rainwater tank as the only non-mains supply (Scenarios 1-3), and (b) Scenario 3 households that have both a rainwater tank and a Class A+ supply (from recycled wastewater or recycled stormwater).

end-use	households with rainwater tanks only		houses with rainwater tanks & Class A+ supply		
	rain water (L/hh/d)	mains water (L/hh/d)	rain water (L/hh/d)	A+ water (L/hh/d)	mains water (L/hh/d)
laundry	43	43	43	--	43
toilet	33	33	0	66	0
external	35	35	0	37	33
other	--	234	--	--	234
total	112	345	43	104	310

An estimate for overall non-residential urban water demand was based on the residential:non-residential split developed for the Gold Coast urban area (Lane *et al.* 2011). Estimates for non-residential use of recycled wastewater or stormwater were taken from the TWCMP study for the Caboolture area, and are detailed below Table 7 and Table 8.

Rainwater Tanks

Estimates for rainwater tank yield and nutrient interception were informed by the stochastic methodology presented in other UWSRA research (Coultas *et al.* 2012). This modelling approach accounts for non-linear scale-up effects, and has been used to demonstrate that water supply planning cannot rely on large scale extrapolation from the results of a single rainwater tank model that uses a 'typical' configuration.

Tank energy use estimates were based primarily on results from the laboratory characterisation of rainwater pump performance (Tjandraatmadja *et al.* 2012), although also considered a number of other recent Australian studies (Retamal *et al.* 2009; WCG 2009). Inventories for tank construction and maintenance were included, following the approach used in a previous study (Lane *et al.* 2011).

Advanced Wastewater Treatment and Reuse

Scenario 2 includes wastewater reuse for agricultural irrigation, while Scenario 3 includes an urban wastewater reuse system. For our analysis, both cases assumed that Class A+ water would be supplied from the existing South Caboolture Water Reclamation Plant (WRP). Modelling of the treatment plant operations was largely based on measured data from the South Caboolture plant, and is described in a companion report (Lane *et al.* 2012a).

[2] It is likely that leaks would affect the end-uses considered in this study, particularly the toilet and outdoor taps. Including this in the modelling would introduce an additional departure from the approaches commonly used in SEQ planning studies, and would be worthwhile. However, further review is required to determine an appropriate set of quantitative assumptions for this issue.

The Class A+ usage profiles (Table 7) were largely based on those used in the MBRC study for the Caboolture area.

For the agricultural reuse scenario, the WRP production modelling was (hypothetically) scaled up to provide the full 8 ML/d as per the TWCMP study assumptions.

For the urban reuse scenario, the demands for industrial use and irrigation of parks/sporting fields were taken from the TWCMP scenario used in the MBRC study. Residential and commercial demands were based on the MBRC estimates for connected population, however were modified to reflect the end-use assumptions described above (Table 6). As a result, the overall Class A+ demand (6.8 ML/d) was substantially less than the total demand of 8.2 ML/d used for the MBRC planning.

Urban usage of Class A+ water was assumed to displace an equivalent amount of mains water[3], except in the case of irrigation sports fields and urban parklands. For these, the calculated mains water offsets followed the assumptions used for the MBRC study (BMT-WBM 2012).

Table 7: Reuse balances for the urban and agricultural wastewater reuse systems scenarios.

User		Use (ML/d)	Mains water displaced	
			(ML/d)	(as % of WW used)
urban reuse scenario	households#	1.8 (27%)	1.8	100%
	commercial#	0.4 (6%)	0.4	100%
	industrial	1.5 (22%)	1.5	100%
	sports fields	1.5 (23%)	1.2	80%
	open space irrigation	1.5 (23%)	0	0%
	total	*6.8*	*4.9*	*73%*
agricultural reuse scenario		8.0	0.0	0%

for A+ supply connected to 17,758 households, and commercial businesses equivalent to 4669 EP

Table 8: Reuse balance for the stormwater harvesting and reuse scenario.

User	Use (ML/d)	Mains water displaced	
		(ML/d)	(as % of SW used)
households#	2.1 (83%)	2.1	100%
commercial#	0.2 (8%)	0.2	100%
industrial	0 (0%)	0.0	100%
sports fields	0.1 (3%)	0.1	80%
open space irrigation	0.1 (6%)	0	0%
total	*2.5*	*2.4*	*94%*

for A+ supply connected to 20,279 households, and commercial businesses equivalent to 2600 EP

Stormwater Treatment (WSUD) and Catchment Management

Models for these options were based primarily on their nutrient removal performance, estimates for which were taken directly from the Caboolture TWCMP study (BMT-WBM 2012). Emissions of fugitive nitrous oxide (N_2O) from the stormwater nitrogen were also included in the modelling, using the assumptions described in Appendix A.

No account was taken of the potential for organic micropollutants and heavy metals in the stormwater. Estimates for infrastructure construction (considered to be minimal), and any ongoing inputs for system implementation or maintenance, were also excluded from the analysis.

Stormwater Harvesting and Use

The TWCMP estimate for stormwater treatment power consumption (600 kWh/kL of product water) was used in this study, given the lack of empirical Australian data on which to base a more informed assumption. Chemicals usage was assumed to be the same (per ML of treated water) as for the South Caboolture WRP. Stormwater micropollutants were excluded from the analysis in this study. Construction inventories were assumed to match those for the South Caboolture WRP.

[3] Table 7 shows a 100% mains water displacement for residential reuse, which differs to the equivalent value shown in a companion study on wastewater recycling (Lane *et al.* 2012a). This reflects the different perspectives used to define the quantity of mains water displacement. This report adopts the perspective used in the MBRC analysis, which defined the displacement as the overall gap between 'typical' mains water demand and the amount required after the adoption of the alternative water supply technologies.

The Class A+ usage profile (Table 8) was based on that used in the Caboolture TWCMP, with the exception of the residential use estimates described above (Table 6).

Mains Water Supply

Mains water supply was assumed to be sourced from sea water desalination - see section 3.2 for further discussion on the rationale for this choice.

The desalination model was based on the Tugun Desalination plant at the Gold Coast, using empirical data for energy use (Poussade *et al.* 2011) and a mix of literature review and expert opinion for estimating chemicals use, waste generation and construction inventories (Lane *et al.* 2011).

Sewage Collection and Treatment

Modelling of the South Caboolture sewage system was largely based on empirical data from the relevant facilities, and is described in a companion report to this study (Lane *et al.* 2012b).

3.1.3. Background Data Collection

Analysis across the life cycle of an urban water system requires that detailed consideration be given to not just the environmental emissions (nutrients, micropollutants, fugitive gases) and extractions (freshwater use) directly associated with the process, but also to the equivalent environmental pressures of a less direct nature. More specifically, this involves the use of detailed estimates for emissions and extractions associated with the manufacture and supply of materials/energy used by the system under study. Such information is described here as 'background data', since it relates to activities that are not directly within the sphere of influence of urban water system managers.

This concept, that the user of such products should pay consideration to the environmental problems caused upstream in the supply chain, is already widely implemented in the 'Scope 2' and 'Scope 3' emissions components of Australian GHG accounting. As is the case with GHG reporting, this can only be implemented in practice if databases on the emissions at the manufacturing and transport stages are readily available.

Data for this study was taken from three such databases. Where possible, the AusLCI database (2012) was used, as this is set to become the premier repository for consistent and transparent data (as required for LCA studies) on Australian products and services. In practice, the AusLCI database is still only in its formative stages, and could only provide inventory data for electricity generation processes. For the use of chemicals, materials and transport services, data was taken wherever possible from the Australian products and services database in the Simapro software package (Grant 2012). Any remaining gaps were filled with European based data from the Ecoinvent database (Frischknecht *et al.* 2007), as this is considered the premier source of international data in the LCA community. Where this was done, some basic modifications were made to increase the data relevance to the Australian context – e.g. electricity use modules in the European data for chemicals manufacture were replaced with the equivalent data for Australian electricity generation.

The challenge is that, where GHG accounting (typically) requires data for only a few substances, there are potentially thousands of emissions of interest in a broader LCA study. Much of the background data available for Australian LCA has been developed in piecemeal efforts, and relies heavily on extrapolation from international datasets. It is therefore likely that the uncertainty associated with the background data used in this study is relatively high.

3.2. The Importance of Data Estimates

The environmental analysis that informed the MBRC TWCMP was primarily focussed on issues related to the ecological health of local SEQ waterways. To this end, a number of different metrics were used over the different stages (options screening; final scenario comparisons) of the plan development. Of these, the two that are most directly compatible with the recommended midpoint approach to LCA are the estimates for discharge of total nitrogen and total phosphorus into receiving waters.

GHG emissions were also included in the analysis of the TWCMP development, both at the options screening stage, and in the final comparison of scenarios. No other 'non-core' environmental issues were canvassed.

Only one quantitative metric was used for social impact analysis in the final TWCMP scenarios evaluation. This involved an estimate of the reductions in potable water demand for each of the scenarios, as a measure of the relative impacts on management of the regions' water supply infrastructure.

Estimates for three of these metrics were generated using the data and assumptions underpinning our study, and compared with the equivalent results obtained from using the default assumptions in the assessments of the TWCMP study (Table 9). The stormwater treatment and catchment management initiatives are not included in this comparison, since we adopted source data directly from the TWCMP study.

Table 9: Changes to the analysis of water supply technologies, caused by the different data sets used for this report (Direct comparison) and the extension of the system boundary to include the full life cycle of the infrastructure (Life cycle contributions)#.

	Direct comparison##			Life cycle contributions^		net change^^	
	potable savings (ML/y)	TN removal (kg/y)	GHG (power use) (t-CO2/y)	TN removal (kg/y)	GHG (all) (t-CO2/y)	TN removal (kg/y)	GHG (all) (t-CO2/y)
Rainwater tanks	-247 (-28%)	-551 (-35%)	-390 (-22%)	-521 (-33%)	+274 (+16%)	-1,072 (-68%)	-116 (-7%)
Wastewater → agricultural reuse	0 (0%)	0 (0%)	+909 (+44%)	-645 (-9%)	+655 (+32%)	-645 (-9%)	+1,564 (+77%)
Wastewater → urban reuse	-494 (-22%)	-248 (-3%)	+754 (+36%)	-1,632 (-21%)	+2,057 (+99%)	-1,880 (-24%)	+2,811 (+135%)

Positive/negative changes indicate that the approach of this study gives a higher/lower result than that estimated in the MBRC TWCMP study. All percentage values shown are referenced to the default result in the MBRC TWCMP study.

Comparison based on improved datasets used in this analysis, but otherwise with an identical analytical scope.

^ Comparison shows the effect of including the indirect life-cycle implications for N emissions to waterways (including emissions happening in different catchments, and those resulting from deposition of N emitted to air); and including scope 1 & scope 3 GHG emission.

^^ Comparison showing the net effect of including both the improved data assumptions, and the full life-cycle scope.

3.2.1. Infrastructure Modelling – Direct Comparisons

The 'Direct comparison' section of Table 9 compares the results obtained in the TWCMP study, with those obtained using the exact same system boundary, but the alternative datasets collected for this report.

Rainwater Tanks

Because our rainwater yield assumptions were so different, our rainwater tank modelling gives lower potable water savings (28% less) and nitrogen diversion (35% less) than those calculated with the assumptions used in the TWCMP study undertaken for the MBRC (Table 9). Other UWSRA research has shown that conventional approaches to rainwater tank modelling can greatly overestimate the effective rainwater yields delivered by the large scale adoption of household rainwater systems (Coultas *et al.* 2012).

Our estimate for the power use *intensity* of rainwater tank supply was similar to that used for the MBRC study. However, our GHG result for the rainwater tank modelling was much lower (-22%). This is because the lower rainwater tank yield translates to a lower pump throughput for systems where the mains backup bypassed the tank itself. In our modelling, we assumed that this would apply to 80% of the rainwater tank installations, with the remaining 20% utilising a tank top-up system for backup mains water. If the top-up system were to be more widely adopted, then the GHG implications of lower than expected rainwater yields would be substantial.

Wastewater Reuse for Agriculture

The MBRC study relied on a generic estimate for the energy requirements to produce Class A+ water, whereas our results were based on actual operating data collected for the South Caboolture WRP. The energy intensity at this plant is much higher than was anticipated (Lane *et al.* 2012a), hence our estimate of power-related greenhouse gas emissions is much higher (+44%) than calculated in the MBRC study.

Wastewater Reuse in Urban Class A+ System

As with the analysis for the agricultural reuse system, the TWCMP study estimates for the urban Class A+ system underestimate the power use required to operate the South Caboolture recycling plant. However, the discrepancy between our results (36% greater) and the default analysis is not quite as large. This is because we assumed an overall product water delivery of 6.8 ML/d, as opposed to the 8.2 ML/d used in the TWCMP study, and therefore the overall energy inputs were also reduced.

The lower Class A+ demand is also reflected in the lower potable savings (-22%) ascribed to our results. A large part of the difference is explained by our assumption that householders will preferentially use mains water for certain outdoor purposes, even when there is an A+ supply available. While that assumption was based on the limited available empirical data, further research would be required for realistic planning estimates to be possible.

3.2.2. Analysis Across the Full Life-Cycle

The life-cycle nutrient modelling indicates that some of the nutrient diversion benefits are partially offset by increases in nutrient flux to waterways occurring at other points in the technology supply chain. The latter are mostly associated with airborne NO_X emissions from electricity generation stacks. However, the NO_X fate modelling that underpins this analysis was based on European assumptions. It is expected that these would overstate the situation in the Australian context (Grant *et al.* 2008). Unfortunately, there are no known equivalent fate metrics for use in Australian LCA.

The differences introduced by the life-cycle GHG modelling are likely to be far more important. Particularly for the wastewater reuse systems, our analysis indicates that the focus on power consumption fails to recognise some notable scope 1 (avoided fugitive emissions for agricultural wastewater reuse and stormwater reuse) and scope 3 (chemicals for wastewater treatment) GHG burdens. This issue is discussed in more depth in a companion report (Lane *et al.* 2012a), and illustrates the potential risk to water utilities in maintaining a narrow GHG focus. Planning decisions based only on Scope 2 GHG estimates could increase the future cost burden for water utilities under a carbon-pricing regime, either through direct payments for Scope 1 emissions, or via costs being passed through the chemicals supply chain.

Overall, the use of ‘improved’ data and a life-cycle scope leads to big differences in the analysis of the different water supply technologies (Table 9). The implications of these changes vary, depending on which indicator and/or technology is under consideration.

3.2.3. Scenario Results

The full TWCMP scenarios were also compared using the improved data collected for this study. The implications for the GHG results are used to illustrate the potential significance this could have in the tradeoffs analysis undertaken as part of any decision making process (Table 10).

A comparison across the results for *potable water savings* and *reduced TN load* shows that in moving from Scenario 1 to Scenario 2, a big nutrient removal benefit is provided. The additional complexity involved in Scenario 3 delivers relatively little additional nutrient removal, but does provide a substantial reduction in the need for potable water supplies. The GHG results from using the default dataset suggests that, while there is a doubling in GHG intensity in moving from Scenario 1 to Scenario 2, there is no additional GHG penalty in adopting the initiatives of Scenario 3. If the decision were to be based on only these three metrics, then the choice might be seen to be between Scenario 1 (low water savings and nitrogen benefits, but low GHG emission) and Scenario 3 (high water savings and nitrogen benefits, at the cost of increased GHG emissions).

However if the improved power use data and full life cycle GHG emissions are factored in, then the interpretation of the tradeoffs might be somewhat different. Firstly, it can be seen that the default datasets significantly underestimate the total GHG intensity of Scenarios 2 and 3, hence the penalty for moving beyond the business-as-usual water cycle configuration will be greater than expected. Secondly, Scenario 3 now has a much higher GHG intensity than Scenario 2, suggesting that Scenario 2 is worthy of consideration as an intermediate solution (medium level water/nutrient benefits, with only a medium level GHG penalty).

Table 10: Comparison of the TWCP scenarios, showing that use of the different datasets would change how the GHG results affect the tradeoffs analysis.

	Potable Water Savings	Reduced TN Load to Caboolture River	Greenhouse Gas Emissions ($kt\text{-}CO_2e/y$)	
	(ML/y)	(t/y)	*default* [1]	*improved* [2,3]
Scenario 1	622	2.0	1.7	1.6 (-6%)
Scenario 2	622	13.6	3.8	5.2 (+38%)
Scenario 3	2,819	14.2	3.7	7.3 (+99%)

[1] Using power use (scope 2) data that is typical of the TWCMP study and many other SEQ water planning studies, and excluding scope 1 or scope 3 emissions.
[2] Using power use (scope 2) models developed for this study, with best estimates included for fugitive gas emissions (scope 1) and the GHG footprint associated with supply of chemicals and infrastructure materials (scope 3).
[3] The % increase values shown in brackets represent the change in GHG result brought about by using the alternative dataset.

3.3. The Importance of System Boundary Selection

For the analysis up to this point, the limited system boundary is also of significant concern.

In constraining the analysis only to local systems (local water supply, wastewater and stormwater infrastructure), the system boundary does match the institutional boundaries of the MBRC. This is consistent with the traditional scope for urban water planning processes, but not so with the notion of *Total Water Cycle* planning as it fails to account for the implications that a choice between these options would have on the operations of the SEQ Water Grid. Scenario 3 would give the lowest demand on the water supply grid, and therefore require the least amount of mains water to be produced. The GHG implications of this avoided mains supply should be included if the analysis is to span the 'Total' water cycle.

Unfortunately, there remains a substantial hurdle to incorporating mains water generation in an appropriate manner. The problem is that there is no clear cut way to choose which water supply technology (or mix of technologies) should be modelled for the mains water offsets. There has been a tendency in urban water systems planning to benchmark against the status quo – often taken as traditional, dam based supplies; or more recently, the current SEQ grid mix. But in the context of growing population and growing overall demand, the status quo provides no insight into what will actually change as a result of decisions that lead to more or less mains water demand.

What should be used in the analysis is the supply source that will change its production rate in the event of a marginal increase or decrease in future demand for mains water. Given the complexity of the SEQ Water Grid, determining the short term or long term marginal supply sources is not a trivial exercise, and will be beyond the capacity of individual planning studies. In order to understand and minimise the GHG implications of the choices they make, urban water planners will need access to a range of modelling forecasts for water grid operations. These would ideally need to reflect both shorter term operational rules for the SEQ Water Grid, and regional policy on the provision of water supply infrastructure to meet the needs of a growing population.

To illustrate the practical implications of such a choice, the basic tradeoffs analysis (Table 10) was extended to consider two alternative approaches to modelling the mains water supply (Table 11). Firstly, a system-wide, averaged, GHG intensity was calculated for the long term water supply scenario modelled in a previous study (Hall *et al.* 2009; Hall *et al.* 2011). Using this value would give Scenario 3 a lower overall GHG intensity than Scenario 2, thanks to the increased potable water savings delivered. With this approach, the decision maker might resolve that the choice comes down to Scenario 1 (lower GHG intensity) vs. Scenario 3 (greater water and nutrient benefits).

Table 11: Comparison of the TWCP scenarios using different system boundaries for the analysis.

	Potable Water Savings (ML/y)	Reduced TN Load to Caboolture River (t/y)	Greenhouse Gas Emissions[1] (kt-CO_2e/y)		
			mains supply excluded [2]	*mains supply included ('avg' grid supply)* [3]	*mains supply included (desal supply)* [4]
Scenario 1	622	2.0	1.6	7.1	15.7
Scenario 2	622	13.6	5.2	10.7	19.3
Scenario 3	2,819	14.2	7.3	9.9	14.2

[1] All GHG scenarios use the improved power consumption data + include estimates for scope 1 and scope 3 emissions.
[2] Identical to the 'improved' scenario from Table 10.
[3] Using a system-wide average value taken from Hall et al. (2009).
[4] Using our data for water supply from the Gold Coast seawater desalination plant.

However, a different conclusion would be reached if seawater desalination is assumed to be the marginal mains water supply source. Because desalination is such an energy-intensive approach to mains water supply, Scenario 3 would then have the most favourable GHG results across all of the three options. If the scope of the decision making were constrained to just these three metrics (*potable water savings*; *reduced TN loads*; *GHG emissions*), then there would be no downsides associated with adopting Scenario 3.

While TWCMP studies will of course need to consider many other issues beyond the three issues considered in Table 9 to Table 11, this simplified example highlights how significant the data and system boundary assumptions might be if they affected an issue that was influential to the decision making process.

3.4. Broadening the Environmental Scope

To provide a broader environmental perspective to the analysis, the same three scenarios were compared using the full set of recommended LCA impact assessment models (Figure 1). This comparison utilises the 'improved' inventory data collected for this study, includes full mains supply in the system boundary, and assumes that this mains water is sourced from a seawater desalination plant.

The comparison shows a complex mix of tradeoffs involved, and therefore no clear cut preference across the scenarios. Interpreting such results, and determining how they might change priorities in the decision making process, will be challenging for urban water planners – particularly if they have no prior knowledge of some of the environmental issues covered in this scope. To overcome this challenge, further information is required on the relative significance of the scale of the different impacts involved. This concept is explored further in Chapter 4.

Direct and Indirect Impact Contributions

Even without such perspectives, this broad spectrum comparison can still provide useful insight to the planning process. Understanding the breakdown of the results would enable planners to identify those elements of their system that are most likely to influence the life-cycle environmental burden. This is illustrated for the Caboolture case study, by identifying the key reasons for differences in the results between Scenario 1 and Scenario 2 (Table 12).

For *Freshwater Extraction* and *Eutrophication Potential*, issues which are normally within the sphere of concern of water system managers, direct process streams make a major contribution to the life-cycle results shown. The wastewater recycling introduced in Scenario 2 reduces the discharge of nutrients into local aquatic ecosystems, hence the differences shown for the *Eutrophication Potential* results. It also reduces the need for direct freshwater withdrawals for crop irrigation, hence the large difference shown for the *Freshwater Extraction* impact category (Figure 1).

For most of the other impact category results, differences in energy use are a major cause of the differences between the two scenarios. This reinforces the conclusion from the previous section that high quality predictions for infrastructure power consumption would be an advantage for water planning studies.

However, these results also identify that accounting for energy use will not, by itself, be adequate for predicting the life cycle environmental burden of water system options. Chemicals use inventories explain the differences in some of the impact category results to a similar, and sometimes greater, degree (Table 12). Construction materials also account for some significant differences, although the study would need to be repeated with higher quality construction inventory data before this result can be confirmed.

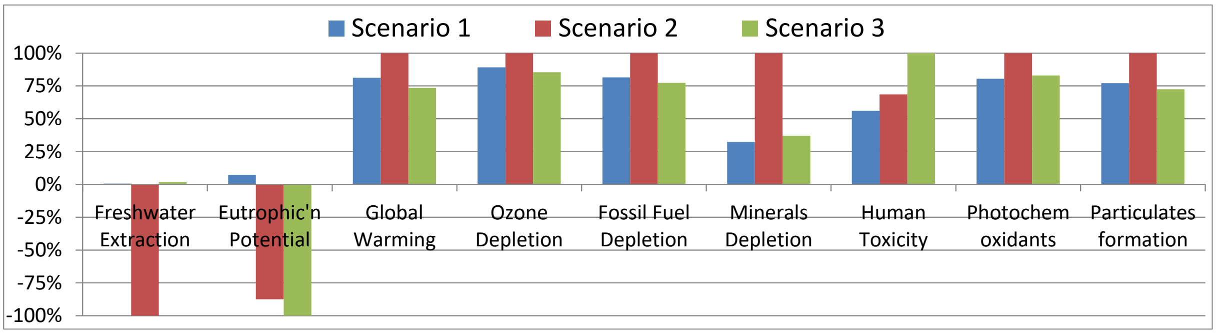

Figure 1: Relative scores for the three scenarios, assessed across the 9 recommended impact models, incorporating mains supply sourced from seawater desalination in the system boundary.

Table 12: Major contributors to the difference in LCA impact results between Scenarios 1 and 2.

		Freshwater Extraction	Eutrophic'n Potential	Global Warming	Ozone Depletion	Fossil Fuel Depletion	Minerals Depletion	Human Toxicity	Photochem oxidants	Particulates formation
overall difference# (s2 - s1)		-2,912 ML/y (-100%)	-13 t PO4e/y (-100%)	3.6 kt CO2e/y (+100%)	2.2 kg CFC-11e/y (+100%)	0.9 kt oil eq/y (+100%)	32 t Fe eq/y (+100%)	38 t 1,4-DCBe/y (+100%)	18 t NMVOC/y (+100%)	8 t PM10 eq/y (+100%)
direct sources^	WW discharge		-78%					-38%		
	irrigation	-100%								
	fugitive gases		-27%		-84%					
indirect sources^	power supply			77%	150%	76%	65%	55%	63%	56%
	chemicals supply			22%	16%	19%	19%	36%	26%	43%
	construct'n				19%	6%	16%	45%	11%	

a positive difference indicates that Scenario 2 (s2) has a higher overall impact than Scenario 1 (s1).
^ these show (in % terms) the major contributors to the overall difference between the two scenarios.

3.5. Broadening the System Scope

The three scenarios developed for the Caboolture TWCMP study excluded any consideration of different secondary wastewater treatment options; hence the sewage collection and treatment systems were excluded from the analysis to this point. Since TWCMP studies in other areas might need to include such options, further analysis is required to understand the significance of using the LCA methodology in this context.

To illustrate the significance of including the wastewater component, Scenario 1 was analysed with all water supply, wastewater and stormwater infrastructure and flows included in the system boundary Figure 2. The results therefore give an indication of the *total impacts* of the water system associated with the urban population considered as the basis for the Caboolture TWCMP study.

The results indicate that the wastewater component (sewage collection and treatment; disposal of treated effluent and biosolids) makes a substantial contribution to many of the life cycle impacts. This concurs with the findings in a previous study on the Gold Coast urban water system (Lane *et al.* 2011).

The implications of this for the TWCMP process are twofold. Firstly, if different wastewater systems were being considered in the options analysis, then data and analytical quality for the sewage collection and treatment systems will be critical to the TWCMP results. Secondly, it raises the question as to whether or not the push for increased STP nutrient removal, taken for granted in the Caboolture TWCMP study, would deliver an overall increase or reduction in life-cycle impacts. These issues are explored further in a parallel report produced by this project (Lane *et al.* 2012b).

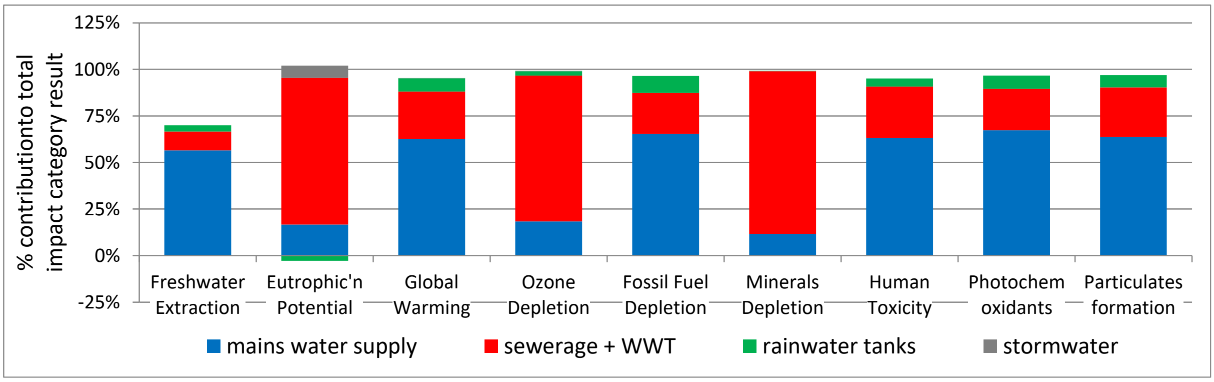

Figure 2: Overall life-cycle impacts of Scenario 1, when the full urban water system (stormwater discharge; sewage collection, treatment and discharge; water supply) are included. Mains water supply is assumed to be from seawater desalination. The sewage/wastewater component makes a substantial contribution to many of the impact results, indicating that this would be an important consideration in TWCMP studies that considered alternate sewage management options.

4. BENCHMARKING OF THE LCA RESULTS

Chapter 3.4 identified that urban water planners would require additional information if they are to incorporate broad spectrum LCA results into the decision making process. This information would need to provide some perspective on the relative significance of various tradeoffs that are identified in an options comparison.

This chapter reviews a number of approaches to providing this perspective, identifying both the benefits and limitations to doing so. This provides an additional contribution to meeting Objectives 2 and 3 of this report.

4.1. Conventional Benchmarking Approaches

In LCA practice, perspective across impact categories is typically provided by comparing case study results against some suitable benchmark. The most commonly used benchmarks are status quo estimates of anthropogenic activity, typically at national (or greater) scales.

The status-quo benchmarking step is described by Equation 2.

$$impact\_score_{NORMALISED} = \frac{impact\_score_{ABSOLUTE}}{impact\_score_{BENCHMARK}} \qquad \textit{Equation 2}$$

This calculation approach normalises the results, providing a consistent set of units across all the impact categories under consideration. For each impact category, the normalised units represent the contribution that the case study makes to the overall status quo, or the amount that the case study changes the overall status quo, for each impact category.

The absolute scale of the normalised results are typically of little interest, since there is often a severe mismatch between the small scale of the case study system boundary (e.g. the Caboolture urban water system), and the much larger scale of the system boundary (e.g. the national economy) used to define the benchmark dataset. However, what is of value with this benchmarking approach is the relative size of the different normalised results. Comparing these relativities provides an indication of which impact category results make the biggest contribution (or change) to the benchmark system. In other words, the bigger normalised results can be considered the most substantial, if considered from the broader perspective that the benchmark provides.

As the scope of life-cycle impact assessment is global in nature, a global benchmark is generally considered to be the most conceptually sound approach to status-quo normalisation (Huijbregts *et al.* 2003). However, because institutional and social priorities tend to be framed more from a national rather than global perspective, a global benchmarking step can lack relevance for decision making at the level of individual organisations. For this reason, benchmarks at the national or supra-national (e.g. the European Union) level are more frequently used in practice.

These two approaches to benchmarking are reviewed below.

Global Benchmarking

For this analysis, the global benchmark was taken from the year 2000 inventory developed for the ReCiPe project (Sleeswijk *et al.* 2008). Impact assessment results were calculated for this inventory, using the same impact models as those used in the scenario comparison of Figure 1. The normalised results were then calculated according to the approach outlined in Equation 2.

This benchmarking step suggests that the tradeoffs associated with *Ozone Depletion* and *Minerals Depletion* have a far smaller influence on the global environmental burden than do those for the other impact categories (Figure 3). No normalised results are recorded for the *Freshwater Extraction* indicator, due to missing information in the source inventory dataset. The normalised results for all other impact categories appear to be of similar significance.

National Benchmarking

The analysis is repeated using a benchmark generated for the Australian economy from the year 2006 (Figure 4). To generate an Australian inventory, we combined a published set of national emissions and extractions data (Foley *et al.* 2009), with pesticide usage data (Lundie *et al.* 2007) and an estimate of total extraction from surface and groundwater systems (National Water Commission 2007).

This benchmarking approach provides a different interpretation on which of the impacts are the most significant. In this case, the *Freshwater Extraction* avoided by the agricultural wastewater reuse scheme has the most substantial effect on the Australian environmental burden. The normalised results for *Eutrophication Potential*, *Global Warming Potential* and *Photochemical Oxidation Potential* are the next most significant.

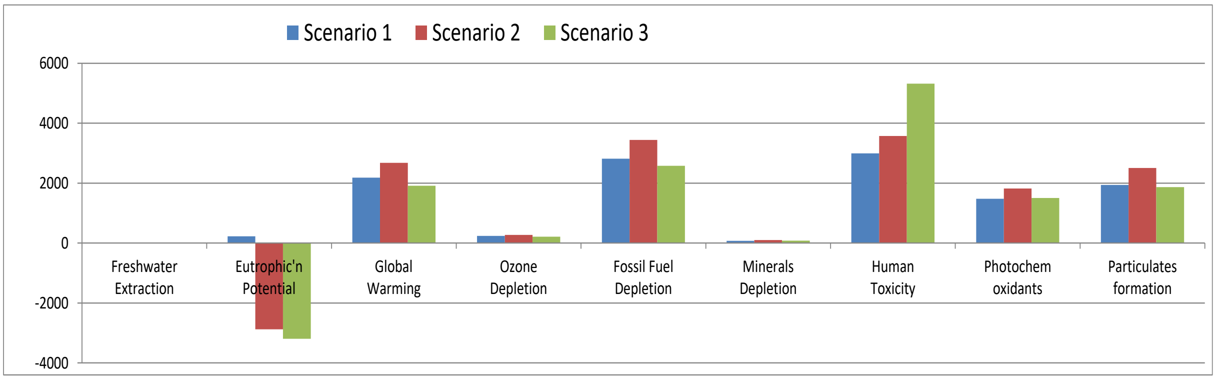

Figure 3: Normalisation of the Caboolture TWCPM scenario comparison against a global benchmark.

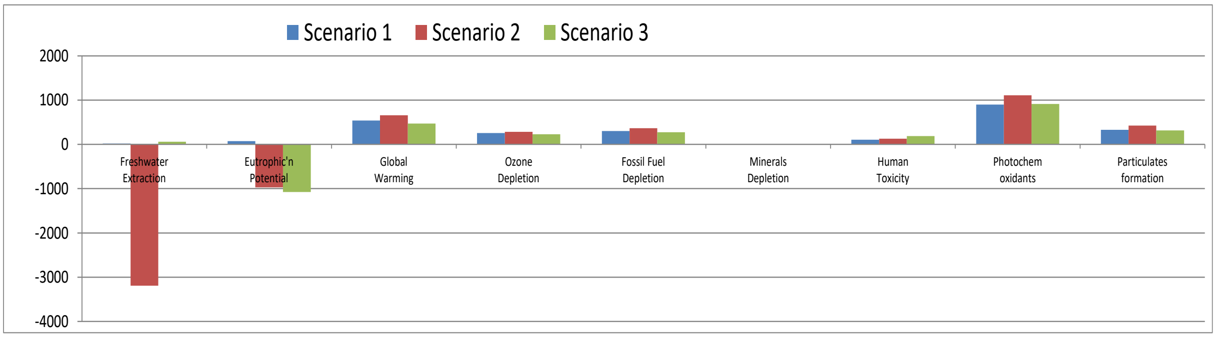

Figure 4: Normalisation of the Caboolture TWCPM scenario comparison against an Australian benchmark.

Comparing the Different Perspectives

In general, the profile of normalised results using the Australian benchmark is quite different to that generated using the global benchmark. The relative significance of some impact results is increased (*Ozone Depletion Potential*, *Photochemical Oxidation Potential*), while for others it is much less than if using the global benchmark (*Fossil Fuel Depletion*, *Minerals Depletion*, *Human Toxicity Potential*, *Particulates Formation Potential*).

The reason for these changes in relative importance can be illustrated by directly comparing the two sets of benchmark LCA results on per-capita basis (Table 13). Not surprisingly, given the relatively high level of economic activity in Australia, the Australian per-capita environmental burden is greater than the world average in nearly all the impact categories considered here. Of particular note are the results for *Fossil Fuel Depletion*, *Minerals Depletion* and *Human Toxicity Potential*, for which the difference between the Australian and global averages is far greater than for the other impact categories. The differences in the ratios shown in Table 13 directly explain the differences in normalised profiles between the global (Figure 3) and national (Figure 4) benchmarking approaches.

Australia's unusually high (per-capita) intensity for *Fossil Fuel Depletion* and *Minerals Depletion* is explained by our dominant role in the global resource extraction industry, and the high degree of coal and mineral exports that underpin our economy. This particular example highlights the problems with using sub-global benchmarking for LCA analysis, particularly for impact categories where the 'problem' is of a global nature. For impact categories that are dominated by an export industry, results normalised with a national benchmark will inherently appear less significant than if the more holistic (global) perspective is used.

The national benchmarking approach makes the resource extraction associated with the Caboolture urban water system appear relatively insignificant (Figure 4), which if used in a decision making context, would diminish the priority given to this issue. Planning outcomes would therefore be more likely to allow increases in materials intensity as an acceptable tradeoff for meeting other objectives. But the associated reductions in resource availability will have a more global effect; hence it could be argued that these small normalised results do not portray the true significance of the water system's contribution to the problem.

Table 13: Per capita comparison of the Global and Australian benchmarks for LCA impact assessment results. Requested replacement image 21/1

		Impacts per capita		ratio of per-capita impacts (Aus / global)
		Global activity	Australian economy	
Freshwater Extraction	(kL/p)		907	
Eutrophication Potential	(kg PO4-e/p)	4	12	3.0
Global Warming	(t CO2-e/p)	7	28	4.1
Ozone Depletion	(g CFC11-e/p)	72	68	1.0
Fossil Fuel Depletion	(t oil-e/p)	1.4	13	9.4
Minerals Depletion	(t Fe-e/p)	0.4	22	48.8
Human Toxicity	(t 1,4-DCB-e/p)	0.1	2	28.1
Photochem oxidants	(kg NMVOC/p)	49	81	1.6
Particulates formation	(kg PM10-e/p)	14	83	5.9

The Problems of Hidden Bias

There is another, more fundamental, difficulty with adopting these conventional approaches to LCA normalisation. This relates to the unknown data quality of the different benchmarks that can be used. For any given benchmark, large differences in quality of the reference values can substantially bias the interpretation of normalised case study results (Heijungs *et al.* 2007).

However, it is very difficult for decision makers (or LCA practitioners) to understand where and how big these biases might be. None of the published LCA benchmark datasets contain uncertainty estimates. Furthermore, the nature and severity of bias will depend not just on the data gaps in the benchmark inventory, but also on how these align with data gaps or uncertainties in the case study inventory (Heijungs *et al.* 2007).

There are many, well-recognised gaps in the Australian benchmark dataset used in this analysis. For example, the estimates for agricultural pesticide use and nutrient losses are considered to be far too low, yet these estimates play a major role in the Australian LCA results (Lundie *et al.* 2007; Foley *et al.* 2009).

These, and other substantial data gaps in the Australian benchmark, are being addressed through ongoing work by the project team. In the interim, caution should be taken when interpreting normalised LCA results that rely on the publicly available Australian normalisation data.

4.2. Organisational Benchmarking Approaches

An alternative approach to case study interpretation would utilise benchmarks that directly relate to organisational performance and/or priorities. Where this is done, the focus is to compare tradeoffs on the basis of their contribution to meeting existing objectives of the organisation in question. An additional benefit is that the data required to generate the "benchmark" are likely to be better understood by those using the LCA results, hence the potential for unrecognised bias in the results interpretation is reduced.

One way to apply this in practice is to benchmark the case study results against status-quo estimates for the organisation.

To demonstrate this example, we generated an overall emissions/extractions inventory for the entire urban water system within the boundaries of the Moreton Bay Regional Council (MBRC) area. This MBRC urban water system dataset utilised a range of available data sources, extrapolating where necessary from the detailed modelling undertaken for the Gold Coast urban area in a previous study (Lane *et al.* 2011).

Benchmarking the Caboolture TWCMP scenario results in this way, gives an indication of the most significant changes that the new infrastructure will make to the overall environmental burden of the urban water utility that manages this region.

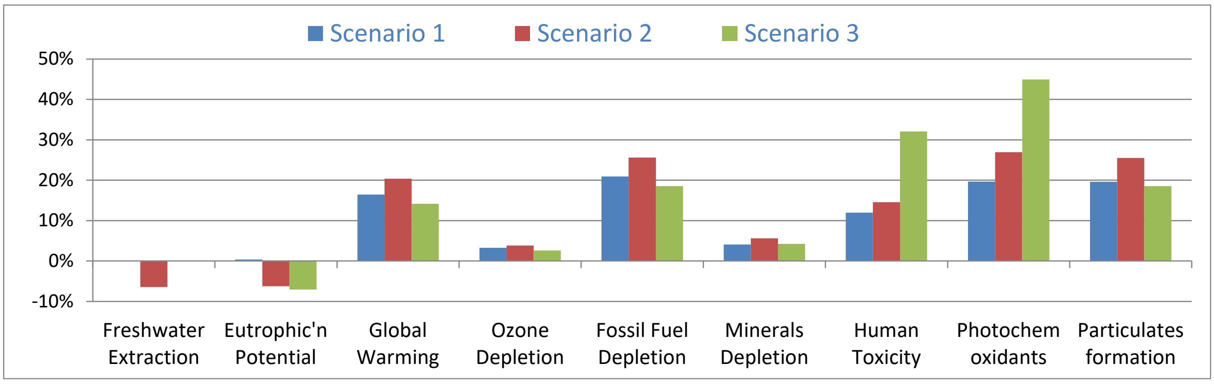

Figure 5: Normalisation of the Caboolture TWCPM scenario comparison against an impact estimate for the urban water system in the Moreton Bay Regional Council area.

Figure 5 shows that the most substantial changes relate to impact categories (e.g. *Global Warming Potential*, *Fossil Fuel Depletion* and *Particulates Formation Potential*) that are not traditionally considered to be the highest priority environmental management concerns for the urban water industry. For this case study, the relatively minor significance ascribed to improvements in *Freshwater Extraction* and *Eutrophication Potential*, is a result of the constraints imposed on the TWCMP planning study.

For the 'externalities' under consideration, a benefit of this benchmarking approach is that it identifies those tradeoffs which might be better addressed by other means. For example, the changes in *Ozone Depletion Potential* appear relatively minor, as the scenarios in question involved little change to the N_2O generation rates from the overall urban water system. It is likely that there would be greater opportunities for N_2O mitigation through changes to sewage treatment or biosolids disposal practices (Lane *et al.* 2012b). As a consequence of this information, decision makers might choose to downplay the significance of the *Ozone Depletion Potential* (and other) tradeoffs identified in the options comparison (Figure 1).

Using the Water Utility 'Status Quo' Approach

The use of this organisational-based benchmarking approach is also demonstrated through a number of other comparisons that focussed on more specific choices of relevance to urban water system planners (Lane *et al.* 2011, 2012a; Lane *et al.* 2012b).

Improvements to the interim MBRC-based dataset, and extending this to include the Sunshine Coast Regional Council urban areas, would provide information that is more directly relevant to the operations of Unity Water. Development of a similar approach for the entire SEQ urban water system might also be beneficial, as it would ensure that the water-supply grid operations are adequately represented in analysis for areas where there will be substantial growth in water demand.

5. CONCLUSIONS

This chapter provides a synthesis of the findings from this report, and the two other reports (Lane *et al.* 2012a; Lane *et al.* 2012b) used to present aspects of the LCA investigation undertaken through this project.

Improving the Quality of Existing Analytical Approaches

The application of LCA to Caboolture case studies has highlighted a number of areas where the TWCMP process could be improved through the use of greater analytical rigour.

Without improved data, it is unlikely that planning studies will be able to accurately predict the effect of direct reuse schemes (rainwater tanks; Class A+ water) on overall mains water demand for new urban developments. While there has been substantial recent progress in understanding the scale and drivers for residential water demand, the research focus in SEQ has, and continues to be, primarily on mains water usage. There is insufficient empirical information available to generate quality assumptions on demands for water from alternative supplies such as rainwater tanks or Class A+ recycling sources. For rainwater tanks, there is an additional challenge in modelling the overall rainwater yield for urban areas that involve large scale adoption of household tanks. The simplified approaches to rainwater yield modelling that are typically used by the urban water industry, are unlikely to be effective in this regard.

Improvements are required in the quality and scope of data used for GHG analysis, if infrastructure planning is to minimise future GHG burdens for water utilities. A reliance on generic power consumption data is unlikely to be sufficient, and more focussed and precise power estimates will be required when there is a complex mix of technologies under consideration. The TWCMP case study analysis presented in this report illustrates how the use of improved data for GHG accounting could significantly change the way that planning options are ranked.

All three reports identify that GHG analysis for the water industry must consider Scope 1 and Scope 3 emission sources, along with the traditional focus on electricity use (Scope 2). The adoption of chemicals-intensive technologies, such as wastewater recycling, could also be introducing substantial exposure to carbon pricing systems for water utilities. With regards to Scope 1 emissions, our report on wastewater treatment (Lane *et al.* 2012b) highlights that research into fugitive emissions of CO_2, CH_4 and N_2O is progressing rapidly in the areas likely to be the biggest source of GHG burden to existing urban water systems. However, further work is required to improve the quality of fugitive gases estimation in planning studies, which are focussed more on options analysis than on overall GHG accounting.

For SEQ planning studies in areas undergoing substantial population growth, robust analysis of mains water supply sources will be critical. The outcomes of options analysis will be extremely sensitive to assumptions on which mains supply source actually changes as a result of planning decisions. There is insufficient information available to determine how best to do this in practice, given the complex nature of the integrated mains supply grid that feeds the SEQ region. The prospect of future policy changes also complicates this issue.

Broadening the Analytical Scope

Neither power use consumption, nor GHG accounting more generally, will provide an adequate proxy for the broader environmental implications of urban water planning. In some decision making contexts, robust analysis of power use, chemicals use, and fugitive gas emissions, might be sufficient. But in other cases, it is likely that the direct use of the full suite of LCA impact assessment metrics would be required to achieve this understanding. Examples of the latter are provided in the two companion reports to this study (Lane *et al.* 2012a; Lane *et al.* 2012b).

Adopting the broad spectrum environmental analysis provided by the LCA methodology would help to highlight a number of issues that are becoming, or could become in the future, concerns of relevance to urban water managers. Three of the most likely candidates are presented in detail in our companion reports to this study (Lane *et al.* 2012a; Lane *et al.* 2012b). Phosphorus recovery is already recognised as an opportunity for the water industry to generate a positive environmental image; however the available LCA metrics are unable to substantiate the significance of such an outcome. In contrast, LCA metrics for ozone layer depletion highlight the significance of N_2O generated by wastewater systems, yet it is not widely recognised in the industry that this could become an issue that water planners are forced to address. The third relates to toxicity assessment, as LCA offers great potential to provide broader perspective on the heightened public concern about the toxicity of organic and metal contaminants in wastewater and biosolids streams. Unfortunately, the available LCA impact models are not currently able to provide meaningful analysis in this regard.

There are a number of other environmental externalities that could also be incorporated into the decision making process through the use of LCA, as summarised in Chapter 2 of this report. Doing so would help illustrate the extent to which water system planning affects environmental issues beyond the traditional concerns of local aquatic hydrology and eutrophication risk. This will, however, be dependent on life-cycle inventories being available for the material/energy inputs and emissions associated with chemicals manufacturing, chemicals transport, and sludge transport. While these can typically be found in Australian LCA databases, there is a need for improved data quality for substances and systems that are of relevance to urban water infrastructure. Chemicals manufacturing is a gap of particular note, because of the potential GHG burden being introduced with the adoption of chemical intensive processes such as wastewater recycling or seawater desalination.

A further challenge with incorporating additional quantitative measures into the decision making process, will be the need for planners to make a judgement on the *relative* importance of these new environmental concerns. One way to do this is to benchmark (or normalise) the results from a case study against some bigger picture reference dataset. However, decision makers should understand that there is no single, definitive way to do this benchmarking step, and a number of different perspectives should be considered.

Conventional benchmarking approaches taken from the LCA domain can provide interesting insight to bigger picture concerns on environmental priorities. However the available datasets needed for this are not sufficiently robust to provide definitive analysis. In this report, we propose an alternative approach that focuses on the performance of urban water systems as a whole. While this does offer a narrower perspective, this benchmarking approach shows promise as a means of analysing tradeoffs for integrated systems, and also for the more focussed decisions that are addressed in our companion reports (Lane *et al.* 2012a; Lane *et al.* 2012b).

Summary of Data Requirements

Along with accurate estimates of water and nutrient flows, LCA analysis of urban water systems will require quantitative estimates to be generated for tens or hundreds of additional parameters.

The minimum data requirements to support the use of broader spectrum LCA in the TWCP process would be as follows:

- quantity of power consumption;
- quantity used, and the distance transported, for all process chemicals;
- distance transported for all sludge byproducts; and
- best estimates for fugitive N_2O, CH_4, CO_2, and NH_3 emissions directly from water-related infrastructure.

If LCA toxicity models were to be included in the analytical framework, then detailed data would also be required on the concentration of contaminants (metal and organic micropollutants) in wastewater and biosolids streams.

APPENDIX A: Estimating Fugitive Greenhouse Gas Emissions

Table 14: Factors for estimating fugitive greenhouse gas emission rates

	Source	Our Assumption		Basis
C	non-biogenic sewage carbon	9	% of total sewage C	Law et al (in prep)
	carbon sequestration			
	- biosolids to landfill		modelled	NGERS (DCCEE 2011)
	- biosolids to agriculture	0.24	fraction of C applied to soils	Brown et al (2010)
CH_4	water supply dams		excluded	insufficient data available to ascertain the implications of marginal changes in dam supply operations
	sewer rising mains conc'n	5	mg-CH_4 /L	various
	fraction of sewer flow through rising mains	100	% of sewer flow	
	STP			
	- aerobic processes		various	Foley et al (2010a)
	- anaerobic processes		modelled	de Haas et al (2009)
	biosolids			
	- interim stockpiling		excluded	
	- to landfill		modelled	NGERS (DCCEE 2011)
	- to agriculture	2.8	g-CH_4 per kg-ds applied	Foley et al (2007)
N_2O	water supply dams		excluded	preliminary research has not yet provided sufficient data for quantitative estimates
	WSUD stormwater treatment	8.5	g-N_2O per kg-ΔN	Foley et al (2010b); Ahn et al (2010)
	stormwater			
	- to freshwater streams	2.4	g-N_2O per kg-N discharged	adopted wastewater emission factors
	- to estuaries	9.4	g-N_2O per kg-N discharged	
	STP secondary treatment	8.5	g-N_2O per kg-ΔN	Foley et al (2010b); Ahn et al (2010)
	wastewater			
	- to freshwater streams	2.4	g-N_2O per kg-N discharged	Foley et al (2007)
	- to estuaries	9.4	g-N_2O per kg-N discharged	
	- to ocean	0.8	g-N_2O per kg-N discharged	
	- irrigation	12.6	g-N_2O per kg-N discharged	
	biosolids			
	- to landfill	6.9	g-N_2O per kg-N applied	de Haas et al (2009)
	- to agriculture	15.7	g-N_2O per kg-N applied	Foley et al (2007)
	fertilisers (avoided)	15.7	g-N_2O per kg-N applied	IPCC (2006b)

CH_4 Generation in Water Supply Dams

The potential for CH_4 emissions from water supply dams is not considered under the IPCC or NGERS greenhouse gas accounting protocols.

There are a small number of examples where dam sourced fugitive greenhouse gas emissions were considered in Australian planning studies, however these focussed only on the CO_2 that would be released from vegetation that gets covered when a dam fills for the first time. However there is ample international (e.g. Delmas *et al.* 2005) and Australian (Grinham *et al.* 2011; Sherman *et al.* 2012) evidence showing that methane generation, as a result of ongoing carbon inputs (e.g. leaf litter, sediment) to the water storage, can also be a significant source of greenhouse gas emissions.

The available data for SEQ water storages (Grinham *et al.* 2011; Grinham 2012; Sherman *et al.* 2012) indicates that emissions can vary substantially depending on water quality characteristics, catchment characteristics, and over time. While there remains considerable uncertainty on the scale of total dam-sourced methane emissions, previous analysis using available information for Little Nerang and Hinze dams, showed that dam sourced CH_4 could add 7% to the overall greenhouse gas burden of the Gold Coast urban water (water supply and wastewater) system (Lane *et al.* 2011). Clearly, this should be an issue that is factored into water supply planning considerations.

However, CH_4 emission estimates from water supply storages were not directly used in this study. The analysis in this report, and the companion study on wastewater recycling (Lane *et al.* 2012a), both considered possible changes to existing dam-sourced water supply rates. However, the available SEQ data on dam methane only provides a guide to total emission rates from overall storage operations. Further review is required before quantitative estimates would be possible for the change in methane generation that might result from changes in the dam operating regimes.

N_2O Generation in Water Supply Dams

N_2O generation from water supply dams is also excluded from the IPCC and NGERS frameworks, and to our knowledge has not been considered in any other studies of Australian urban water systems. The potential for dam sourced N_2O generation is not well understood, either in Australia or internationally.

Research currently underway has confirmed that N_2O is being emitted from some SEQ water storages (Grinham 2012). Further work is required to quantify overall N_2O generation rates, understand the fundamental mechanisms that might result in N_2O emissions, and determine the net anthropogenic contribution to N_2O emissions created by the construction and operation of SEQ dams. However, initial indications from the available data are that dam sourced N_2O is unlikely to be as significant as other fugitive greenhouse gas sources across the urban water system.

N_2O Emissions Associated with Stormwater Management

Neither the IPCC nor Australian NGERS accounting protocols require consideration of N_2O emissions from stormwater sourced nitrogen. Nor are N_2O emissions a priority focus of Australian stormwater related debate. We are not aware of stormwater-related fugitive greenhouse gas estimates being included in any greenhouse gas audits or life-cycle assessment studies on urban water systems.

In the absence of more specific guidance in the literature, we have assigned N_2O emission factors for stormwater discharge equal to those used for wastewater discharge.

Our literature review did not cover the possibility of N_2O generation from bio-physical stormwater treatment systems. On the assumption that it would represent a worst case scenario, we used an N_2O emission factor for stormwater treatment nitrogen removal (8.5 g-N_2O/kg-N removed) equivalent to that used for sewage treatment plant operations.

CO_2, CH_4 and N_2O Emissions Associated with Wastewater Treatment

Detailed summaries on the state of the science, and the assumptions used in this project, are provided in the companion report on wastewater treatment (Lane *et al.* 2012b).

GLOSSARY

General

AWTP	advanced wastewater treatment plant
BAC	biological activated carbon
Class A+	recycled water classification as defined by the Qld state government (2008b)
GHG	greenhouse gas(es)
IPR	indirect potable reuse
LCA	Life Cycle Assessment
LCI	life cycle inventory
LCIA	life cycle impact assessment
MBRC	Moreton Bay Regional Council
MF	micro-filtration
RO	reverse osmosis
SEQ	South East Queensland
STP	sewage treatment plant
TEQ	toxicity equivalent concentration
TWCM	Total Water Cycle Management
TWCMP	Total Water Cycle Management Planning
UF	ultra-filtration
WRP	water recycling plant
WSI	Water-stress Index

LCA Impact Categories

EP	Eutrophication Potential
FETP	Freshwater Ecotoxicity Potential
FFD	Fossil Fuels Depletion
FWE	Freshwater Extraction
GWP	Global Warming Potential
HTP	Human Toxicity Potential
MD	Minerals Depletion
METP	Marine Ecotoxicity Potential
ODP	Ozone Depletion Potential
PMFP	Particulate Matter Formation Potential
POFP	Photochemical Oxidants Formation Potential
TETP	Terrestrial Ecotoxicity Potential

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