Life Cycle Assessment Perspectives on Wastewater Recycling

Joe Lane[1,2], David de Haas[1] and Paul Lant[2]

December 2012

Urban Water Security Research Alliance
Technical Report No. 86

Queensland Government
National Research FLAGSHIPS Water for a Healthy Country
CSIRO
Griffith UNIVERSITY
THE UNIVERSITY OF QUEENSLAND AUSTRALIA

Urban Water Security Research Alliance Technical Report ISSN 1836-5566 (Online)
Urban Water Security Research Alliance Technical Report ISSN 1836-5558 (Print)

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.

For more information about the:

UWSRA - visit http://www.urbanwateralliance.org.au/
Queensland Government - visit http://www.qld.gov.au/
Water for a Healthy Country Flagship - visit www.csiro.au/org/HealthyCountry.html
The University of Queensland - visit http://www.uq.edu.au/
Griffith University - visit http://www.griffith.edu.au/

Enquiries should be addressed to:

The Urban Water Security Research Alliance
PO Box 15087
CITY EAST QLD 4002

Ph: 07-3247 3005
Email: Sharon.Wakem@qwc.qld.gov.au

Project Leader – Shiroma Maheepala
CSIRO Land and Water
HIGHETT VIC 3190

Ph: 03-9252 6072
Email: Shiroma.Maheepala@csiro.au

Authors: 1 – University of Queensland, Advanced Water Management Centre; 2 - University of Queensland, School of Chemical Engineering.

Lane, J.L., de Haas, D. and Lant, P.A. (2012). *Life Cycle Assessment Perspectives on Wastewater Recycling*, Urban Water Security Research Alliance Technical Report No. 86.

Copyright

Disclaimer

Cover Photograph:

Description: South Caboolture Water Reclamation Plant (*left*), South Caboolture Sewage Treatment Plant (*right*).
Photographer: Cristoph Ort (2009)

ACKNOWLEDGEMENTS

This research was undertaken as part of the South East Queensland Urban Water Security Research Alliance, a scientific collaboration between the Queensland Government, CSIRO, The University of Queensland and Griffith University.

Julien Reungoat provided valuable data and feedback on the analytical approach. The report benefited from useful guidance on toxicity analysis provided by Beate Escher and Janet Tang.

Particular thanks go to the staff of Unitywater (Andrew Sloane, Phil Wetherall, Niloshree Mukherjee), the Moreton Bay Regional Council (Lavanya Susarla), and BMT WBM (Nicole Ramilo, Tony Weber). Without their contributions of time, data and insight, this project would not have been possible.

The report also relied on contributions from other UWSRA researchers and research projects, namely Murray Hall, Esther Coultas, Luis Neumann, Shiroma Maheepala, Grace Tjandraatmadja, Cara Beal and Rodney Stewart.

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.

Chris Davis
Chair, Urban Water Security Research Alliance

CONTENTS

LIST OF FIGURES

LIST OF TABLES

EXECUTIVE SUMMARY

The Urban Water Security Research Alliance (UWSRA) 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 SEQ.

This report is one of three produced to meet that goal. In this particular report, the focus is on urban wastewater recycling systems. A number of large recycling schemes have been implemented in SEQ over the last 10 years, and wastewater recycling options continue to feature prominently in planning studies and public debate.

The first two objectives of this report were to: (1) quantify the environmental implications of Class A+ wastewater reuse; and (2) identify any broader benefits and constraints in using the LCA methodology for this analysis. To this end, the report specifically considers the recycling of wastewater generated in the Caboolture sewage catchment. The focus is both on the ozone-BAC technology currently in use at the South Caboolture Water Reclamation Plant, and the direct non-potable reuse of Class A+ water. The third objective of the report was to compare different treatment technologies and reuse systems relevant to urban water planning in SEQ. This was addressed by comparing the greenhouse gas (GHG) footprints of three different combinations of treatment technology (ozone-BAC; UF; MF-RO) and urban reuse type (Class A+; indirect potable reuse).

The GHG footprint of the ozone-BAC technology is lower than for MF-RO based treatment, but much higher than suggested by the available data for the UF based membrane treatment utilised at the Pimpama-Coomera Class A+ treatment plant. However, this comparison of technologies cannot be confirmed without access to more disaggregated data collection at the Class A+ treatment plants. In particular, there is a need to understand electricity consumption for Class A+ product water distribution – this is an often overlooked aspect of water supply options analysis.

The power use implications of water supply offsets are potentially far more substantial than those associated with the treatment technologies. However, generating robust, quantitative estimates on this aspect will not be possible without substantial improvements in analytical technique and data availability. For direct (Class A+) reuse systems, estimating the scale of water supply offsets will require greatly improved end-use data. Interactions between rainwater tank and recycled wastewater supplies should also be considered carefully when benchmarking Class A+ systems against alternative approaches to urban water supply. For planners to develop a realistic understanding of the true environmental implications of wastewater recycling, they will need to understand what changes might be caused to the mix of mains water supply sources that feed the SEQ water grid.

While important, power use assessments are not sufficient to identify the relative GHG footprints of different recycling options, nor for understanding the broader suite of life-cycle environmental impacts associated with wastewater recycling schemes. Data on chemicals use will also be critical, as could estimates for fugitive gas emissions under some circumstances.

Consideration of a more diverse set of LCA based impact categories could be useful, particularly if more diverse wastewater recycling options are under consideration. LCA toxicity analysis could play a particularly valuable role, by providing a range of broader perspectives on the high profile concerns over organic micropollutants in recycled wastewater streams. Unfortunately, the available LCA toxicity models aren't currently able to provide meaningful analysis in this regard. Nonetheless, this project has identified a number of opportunities to address this gap through further research.

1. INTRODUCTION

Project Context

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

LCA was chosen because of its capacity to provide rigorous, quantitative analysis using (a) robust and transparent approaches to system boundary definition; and (b) science-supported impact assessment models (Bauman *et al.* 2004; Schnoor 2009). LCA has previously been used to analyse various aspects of the urban water system in SEQ (de Haas *et al.* 2008; de Haas *et al.* 2009; Foley 2009; Lane *et al.* 2011).

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 standalone 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* (Lane *et al.* 2012b) provides the parent study for the LCA investigation, and synthesises findings across all three reports. It also directly applies LCA to a specific TWCMP 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.
>
> *Application of Life Cycle Assessment to wastewater systems planning* (Lane *et al.* 2012a) provides a more detailed investigation into conventional urban sewage management systems, addressing a key gap identified in the parent study (Lane *et al.* 2012b) on the Caboolture TWCMP.
>
> *Life Cycle perspectives on wastewater recycling* (this report) 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.

Conflicting Priorities for the South Caboolture AWTP

The South Caboolture Water Reclamation Plant (WRP), operated by Unitywater, is one of the largest advanced wastewater treatment plants (AWTPs) in SEQ. However, it has been substantially underutilised in terms of wastewater reuse. Operation of this plant has traditionally been characterised by high throughput (average 7.5 ML/d) but low actual use (~ 2 ML/d) of the Class A+ product water[1].

Until now, the primary role for the AWTP has been to remove additional nutrients from secondary treated effluent prior to discharge to the Caboolture River. The first steps of the treatment train (denitrification; dissolved air flotation/filtration) can be operated independently when nutrient removal is the only requirement, bypassing the second module (ozonation; biologically activated carbon; disinfection) that is required to reach the Class A+ water quality standard.

---

[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.

However, the nutrient removal role of the South Caboolture AWTP is no longer critical. Recent changes at the South Caboolture sewage treatment plant (STP) mean that the STP will now be capable of producing licence-quality effluent without the need for any downstream polishing.

This allows Unitywater greater flexibility in how it chooses to operate the AWTP. While they might choose to continue the AWTP operation as before, there are other conflicting drivers that might also influence the longer term operational strategy.

On the one hand, the question might be raised as to whether operation of the South Caboolture AWTP should continue, given the low demand for Class A+ water at present. Industry forums often raise the concern that the cost and regulatory requirements associated with providing Class A+ water for reuse act as a substantial deterrent to water utility interest in such systems.

On the other hand, any move to mothball or constrain Class A+ water production would conflict with the strategic direction pursued by the recent Total Water Cycle Management Planning (TWCMP) study for the Caboolture area (BMT WBM 2012). The TWCMP study recommended that wastewater recycling play a major role in catering for population growth in the Caboolture urban area. Their analysis assumes that population growth in the region will generate sufficient demand for Class A+ water to take up the full supply capacity of the existing South Caboolture AWTP.

A Growing Interest in Micropollutant Removal

Recent monitoring of the organics removal achieved by the ozone-BAC treatment train (Reungoat *et al.* 2012) has generated additional interest in the South Caboolture AWTP. Their analysis indicates that the South Caboolture WRP product water would likely meet the requirements for augmentation of drinking water supplies as proposed by the *Australian Guidelines for Water Recycling* (NHMRC and NRMMC 2011). This complements a separate study which showed that, for certain high profile chemicals-of-concern, ozone-BAC treatment could achieve similar levels of organics removal as reverse osmosis membranes (Lee *et al.* 2012).

While not conclusive, these findings do support the view that ozone-BAC treatment might be a suitable technology for indirect potable reuse (IPR) systems. This would be of particular interest for inland locations, as reverse-osmosis processes generate a concentrated brine byproduct that can be problematic to dispose of away from coastal areas. The high quality product water from the ozone-BAC treatment process could also attract attention if the future brings an increased focus on chemical health risks in Class A+ wastewater recycling schemes.

Assessing the Environmental Implications of Ozone-BAC Treatment

Given the growing interest in the South Caboolture plant and its ozone-BAC treatment train, the **first objective** of this report was to:

1. *Quantify the life-cycle environmental implications of Class A+ wastewater recycling from the South Caboolture AWTP.*

To support this, the **second objective** was to:

2. *Identify broader benefits and constraints associated with using the Life Cycle Assessment (LCA) methodology for the analysis of wastewater recycling systems.*

Chapter 5.1 addresses Objectives 1 and 2 by providing broad spectrum environmental analysis of wastewater reuse from the South Caboolture AWTP, across a range of environmental and human health impact categories. While intended to inform any debate by Unitywater on future operation of the treatment plant, the analysis in this report is based on hypothetical wastewater conditions that don't strictly match the current situation at the South Caboolture plant. However, these conditions match the approach used in the Caboolture TWCMP study (BMT WBM 2012), and are more likely to be relevant to planning studies in other jurisdictions.

Chapter 5.2 furthers the discussion on Objective 2, by focussing in more detail on the toxicity results generated using the LCA models. In particular, the LCA toxicity results are compared and contrasted with the detailed bioanalytical benchmarking undertaken for the South Caboolture WRP by Reungoat *et al.* (2012).

The Need for Robust Comparisons

A variety of other treatment technologies and reuse systems are in use, or under consideration, for wastewater recycling in SEQ. Scrutiny of different recycling options is likely to continue as SEQ councils implement the total water cycle planning framework that is now required under state government legislation. Councils and utilities will therefore require high quality information on a range of recycling approaches if they are to deliver sound recommendations on the options available to them.

However, there is very little information available to support robust comparison across the range of possible recycling approaches. Additional analysis is required to inform debates on the relative pros and cons of the different possibilities. **Objective 3** for this report was therefore to:

3. *Compare different treatment technologies and reuse systems relevant to urban water planning in SEQ.*

Chapter 1.1 addresses Objective 3 by directly comparing three different scenarios:

(1) the South Caboolture Class A+ scenario using ozone-biological activated carbon (BAC) treatment technology (as developed under *Objective 1*);

(2) an equivalent Class A+ reuse system, but using a membrane based treatment technology; and

(3) a hypothetical indirect potable reuse scenario using reverse osmosis and advanced-oxidation treatment.

The analysis in Chapter 1.1 is limited to a comparison of the greenhouse gas (GHG) footprints for each system, focussing on differences related to either the choice of treatment technology and/or the type of reuse system. Empirical data from actual SEQ systems is used in each case. However the analysis assumes that each of the treatment plants is located at the South Caboolture WRP site, in order to avoid any influence of differences in location.

2. CASE STUDY OVERVIEW

2.1. STP Operations

Actual STP operations were not modelled as part of the system boundary. Instead, the scenarios developed for this report all assume that the South Caboolture STP is producing treated effluent with 2.5 mg/L total nitrogen and 0.3 mg/L total phosphorus, the design conditions for the STP operation following the recent capacity and nutrient removal upgrade. Each of the three AWTP technology models (described below) was adjusted to account for this low nutrient effluent being used as the feedstock. Any STP processing required to treat return flows (e.g. backwash) from the AWTPs was excluded from the analysis.

It should be recognised that, even for modelling of the ozone-BAC system currently installed at the South Caboolture AWTP, these effluent feed conditions represent a hypothetical scenario. With the current configuration at the South Caboolture site, the AWTP runs in parallel to the tertiary nutrient removal unit operations that were installed with the STP upgrade. However, the low nutrient concentration conditions were used for the modelling in this study, as this would be more representative of the situation facing decision makers who are considering the use of wastewater recycling at one of the other advanced nutrient removal STPs in SEQ.

Each scenario assumed that the transfer of secondary treated effluent to the AWTP avoided an equivalent amount of secondary treated effluent discharge to the Caboolture River estuary (Figure 1 and Figure 2). In other words, it was assumed that the only other disposal pathway for the STP is to receiving waters. This is the case at South Caboolture, where there is no history of distributing Class B water (secondary treated effluent) for direct irrigation use.

2.2. South Caboolture Ozone-BAC System

The ozone-BAC treatment train of the South Caboolture WRP actually involves two distinct modules: Module 1 (denitrification; dissolved air flotation/filtration) provides preliminary organics and nutrient removal; while Module 2 (ozonation; biological activated carbon; disinfection) provides the necessary treatment steps to reach the Class A+ water quality standard.

Inflow to the ozone-BAC treatment process was set at 8.3 ML/d, in order to meet the defined Class A+ product water demand of 6.8 ML/d. The difference is the filter backwash that is recycled to the adjoining STP.

Following the assumption that the feedstock will be at low effluent nutrient concentrations, the dedicated denitrification step that is currently employed at the South Caboolture WRP was excluded from the modelling.

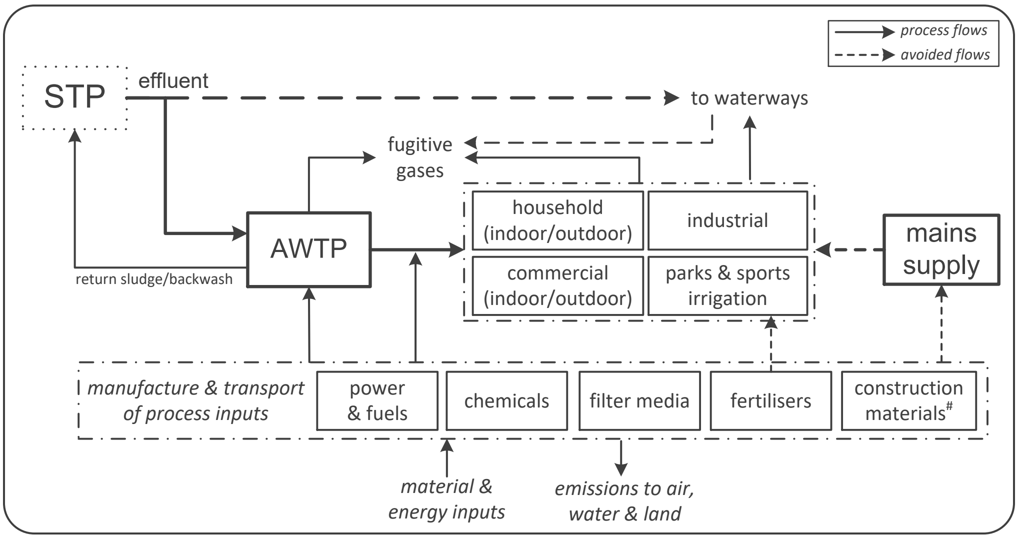

Figure 1: Life cycle system boundary for analysis of the Class A+ reuse systems (#construction materials are excluded from greenhouse gas analysis in section 1.1).

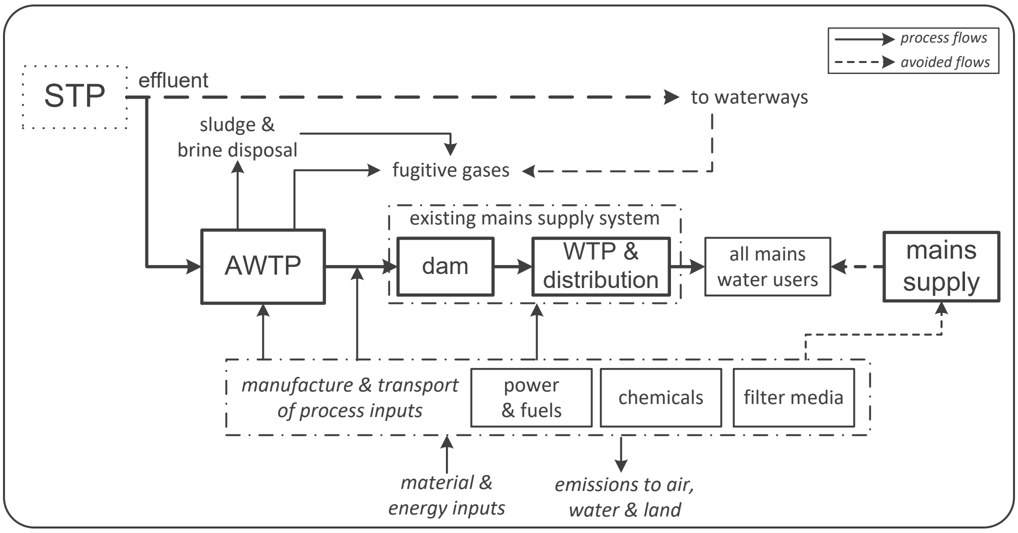

Figure 2: Life cycle system boundary for analysis of Indirect Potable Reuse (IPR) system.

The Class A+ reuse scenario assumes that 100% of the product water is distributed directly to a mix of different urban end users (Table 2). The likelihood that each end use type would displace some alternative water supply was included in the modelling.

2.3. Alternative Reuse Systems

For the GHG analysis, the South Caboolture Class A+ system was benchmarked against two (hypothetical) alternative approaches to wastewater recycling. In both these cases, it was assumed that the alternative technology is implemented at the South Caboolture site. Feedstock related conditions (wastewater quality; transfer pumping requirements) were assumed to be the same for each of the three systems.

An **alternative technology for Class A+ reuse** was modelled on the UF-based technology used at the Pimpama-Coomera AWTP. The total inflow (17.3 ML/d) and product water generation (17.1 ML/d) for this scenario were much greater than for the Class A+ scenario based on the ozone-BAC technology. However, the breakdown (in % terms) of the product water distribution, and the end-use destinations, were the same (Table 2).

The third scenario consisted of an **indirect potable reuse (IPR) system**, based on the option considered (but not recommended) through the Total Water Cycle Management Planning (TWCMP) study recently undertaken for the Caboolture region (BMT WBM 2012). The IPR scenario utilised North Pine Dam as the intermediate storage for the AWTP product water, with the product water subsequently distributed through the existing mains water supply network (Table 3).

2.4. System Boundary

2.4.1. LCA of the South Caboolture Reuse System

Functional Unit

The review of LCA impact results (across the full set of impact categories) was applied to one year of operation of the South Caboolture Class A+ reuse scenario.

System Expansion

As the South Caboolture product water was assumed to reduce the need for mains water and household rainwater tank supplies, production of the displaced amounts from these two sources was credited to the system inventories. For irrigation of the public areas, the possibility that Class A+ water would reduce the rate of freshwater use from local streams was excluded. This was premised on the assumption that none of the non-residential irrigation users have access to direct creek withdrawals.

Fertiliser offsets were allocated to the wastewater nutrients applied to sports fields, on the premise that the managers of these facilities would have both the financial incentive and capacity to adjust their synthetic fertiliser use. Fertiliser offsets for other irrigation users were excluded.

Life Cycle Stages

The system boundary for this analysis included operations of the system, and the construction of the infrastructure required for the treatment plant and distribution networks. The construction inventories were annualised on the basis of estimated lifespans for each major infrastructure item. The disposal of the infrastructure was not modelled, as this is typically found to make a negligible contribution to LCA studies of urban water systems (Lane *et al.* 2011).

Power Supply Sources

Power supply models were based on the average power generation mix for the Queensland state grid. While this provides some consistency with conventional GHG accounting frameworks used by the Queensland water industry, we highlight in this report some constraints to this approach.

2.4.2. Greenhouse Gas Comparison of Alternative Treatment Systems

Functional Unit and System Expansion

For this analysis, the recycling scenarios were compared in two different ways.

The **first comparison** used a *cradle-to-gate* approach, in which the technologies were compared on the basis of 1 ML of product water generated at, and distributed from, the treatment plant. This was used primarily to highlight key differences in (a) the treatment technologies under consideration; and

(b) the pumping requirements for the different approaches to product water use. All downstream impacts of the product water distribution were excluded.

The **second comparison** used a *cradle-to-cradle* approach for the product water, comparing the scenarios on the basis of 1 ML of mains water that is displaced. This provides a more holistic analysis of the life-cycle of the product water, and therefore gives a more meaningful comparison of the different reuse systems.

Displaced rainwater tank operation was included as a credit for the Class A+ scenarios. However, mains water production credits were not included in this particular analysis. Instead, the results for each scenario were normalised to give a common basis of 1 ML of mains water displaced.

Displaced fertiliser use was modelled as described in the previous section.

Life Cycle Stages

The quality of the infrastructure construction inventories available to this project was not sufficient to support the comparative analysis. The infrastructure construction stage was therefore excluded from the scenario models in this section.

Power Supply Sources

Power supply systems were modelled as described in the previous section.

3. DATA COLLECTION

3.1. South Caboolture Ozone-BAC Treatment and Class A+ Reuse

3.1.1. Process Operations Data

An operations model for the ozone-BAC treatment plant was developed from three years of process data (2009-2011) provided by Unitywater.

Total feed flow (8.2 ML/d) was set at the level required to meet the demand for Class A+ reuse (6.8 ML/d), allowing for the substantial sand filter backwash flow that is returned to the STP (1.4 ML/d). The Class A+ reuse balance is discussed further in the following section.

A simplified power use model was developed from historical data, and used to predict total power use at the throughput of the chosen scenario. Data for this model was taken from periods in which the entire treatment process was in use, i.e., excluding those periods when the module 2 operations (ozonation-BAC-disinfection) were being bypassed. The power use model distinguishes fixed (time dependent) vs. flow specific power demands, so as to improve the overall modelling for the specific flow rates used in this study.

The South Caboolture AWTP does not collect disaggregated power use data for their distribution system, hence estimates for this were based on other sources. In the absence of local data, power use for pumping to the residential/commercial Class A+ network was set to equal the equivalent estimate (436 kWh/kL) for the Pimpama-Coomera Class A+ system (Lane *et al.* 2011). For distribution to the other end users, power use was calculated with a simplified hydraulic model (from 140 kWh/kL to 310 kWh/kL), assuming that each of these are supplied by a discrete piping system from the AWTP product water tanks.

Chemicals use estimates were based on an interrogation of the historical plant data, but with the ethanol dosing excluded on the premise that the denitrification function of the AWTP is not required. For our operations model, aluminium sulphate and sodium hydroxide use (for flocculation and pH correction) make up 75% of the total mass of chemicals used for treatment operations. Sodium hypochlorite estimates for final product water disinfection assumed a dosing rate of 60 mg/L, with an additional quantity allocated to the intermediate dosing points within the treatment process.

Fugitive gas emissions (CO_2, N_2O, NH_3) across the system followed the modelling approach used in previous analysis on secondary and advanced wastewater treatment systems (Lane *et al.* 2011), and the basis for these are described in detail in a companion report to this study (Lane *et al.* 2012a).

3.1.2. Class A+ End Use Profiles

Residential

Our residential reuse analysis uses an average household end-use profile (Table 1) that is described further in a companion report (Lane *et al.* 2012b). This is based on the assumptions used for the TWCMP study recently undertaken for the Caboolture region (BMT WBM 2012), supplemented with additional detail from recent end-use analysis undertaken in SEQ (Beal *et al.* 2011).

The residential wastewater reuse model assumed that all houses connected to the Class A+ supply would be newly constructed, detached dwellings which would be subject to the requirements of QDC MP4.2 (QG 2008a). This building code stipulates that new households install either (a) a rainwater tank connected to outdoor, toilet and laundry uses; or (b) some alternative supply that can displace 70 kL/y of mains water supply. Consistent with the TWCMP study, we assume that residential use of Class A+ water is only for outdoor purposes and toilet flushing. This has two implications for the wastewater reuse modelling.

Firstly, we assumed that Class A+ supply would displace the need for rainwater tank supply to outdoor and toilet demands, since rainwater tanks have become (in effect) the default supplier of water to these uses in new SEQ housing. However, under conventional conditions, rainwater yield by tank systems is invariably less than the total demand on them, and mains supply is periodically required to make up shortfalls. The rainwater yield was set at 50% in our scenario (see below), meaning that for the default household with toilets and outdoor uses provided by a rainwater tank, 50% of the demand is actually sourced from the mains network. As a consequence, the residential supply of Class A+ water will still result in the direct displacement of mains supply. The net effect on household mains water balances depends on a complex set of interplays between rainwater tanks and mains water use (for the default scenario), and Class A+ water and mains water use (for the case where Class A+ water is supplied).

Table 1: Water use supply and demand balances for (a) a 'default' household, and (b) households of the two 'Class A+' scenarios.

Use		base demand	default household (rainwater tank supply)		Class A+ scenario (rainwater tank & Class A+ supply)		
			rain water	mains water	rain water	A+ water	mains water
laundry	(L/hh/d)	87	43	43	43	--	43
toilet	(L/hh/d)	66	33	33	0	66	0
external	(L/hh/d)	70	35	35	0	37	33
other	(L/hh/d)	234	--	234	--	--	234
total	(L/hh/d)	457	112	345	43	104	310
	(kL/hh/y)	167	41	126	16	38	113

Secondly, rainwater tank supplies for laundry use would still be required to satisfy the QDC requirements. The maximum possible A+ supply to toilet (66 L/hh/d; 24 kL/hh/y) and outdoor taps (70 L/hh/d; 26 kL/hh/y) would be 50 kL/hh/y. This is less than the QDC requirement of 70 kL/y from alternate household sources. Furthermore, we assumed that householders with a Class A+ connection would still meet 47% (33 L/d) of their outdoor water use from taps connected to mains water supply. This latter assumption is based on a combination of survey and end use monitoring at households in the Pimpama-Coomera Class A+ reuse system (Stewart 2011; Willis *et al.* 2011). As a result, the average household Class A+ use in our scenario was only 38 kL/y.

Non-Residential

For all non-residential users of Class A+ water in our scenario, we assumed that the only alternative would be mains water. The demand profile for these uses, and the effective mains offsets that would be delivered (Table 2), were taken from the assumptions used in the Caboolture TWCMP study (BMT WBM 2012). The low levels of mains water displacement ascribed to irrigation of parks (sports fields and open space) also follows the approach used in the TWCMP study. This reflects the expectation that, with Class A+ water available, total parkland irrigation rates would increase beyond that which would happen if mains water were the only option.

Table 2: End use balance for Class A+ product water distribution. The volumetric balance was used for modelling of the South Caboolture (ozone-BAC) system. The % breakdown was also used for an alternative Class A+ scenario using UF-based treatment.

User	Use (ML/d)	Mains water displaced		Rainwater displaced	
		(ML/d)	(as % of WW used)	(ML/d)	(as % of WW used)
households	1.8 (27%)	0.6	34%	1.2	66%
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	3.7	55%	1.2	18%

3.1.3. Wastewater Micropollutant Data

Organic micropollutant concentrations for the plant influent and product water were based on data collected at the South Caboolture treatment plant (Reungoat *et al.* 2012). That study showed little difference in the concentration of target compounds before and after the denitrification step. We therefore assumed that excluding the denitrification step from the AWTP operations would not affect the potential for organic micropollutants removal.

Indicative metals removal efficiencies for the AWTP were based on a single sample collected at the South Caboolture WRP, supplemented with data from the Pimpama advanced wastewater treatment plant.

3.2. Alternative Class A+ Treatment Technology

The operations model for the Pimpama-Coomera AWTP was taken from the 'Design' scenario developed for a previous study (Lane *et al.* 2011), but modified to match the nutrient concentrations in the hypothetical South Caboolture STP effluent feed. AWTP product water nutrient concentrations were not changed from that previous study. As for the scenario using ozone-BAC technology, the analysis excluded the additional STP processing required to treat the recycle stream from the AWTP operations.

The original flow rates (17.3 ML/d inflow; 17.1 ML/d product water) were used for the modelling in this analysis, with the product water allocated to the same end-use profile (on a % basis) as for the ozone-BAC scenario. For the normalised results, this eliminates any difference between the two Class A+ scenarios relating to the end use of the product water. However, by modelling the Pimpama-Coomera plant at the higher throughput (17.1 ML/d rather 6.8 ML/d), possible economies of scale were not considered in this analysis.

3.3. Indirect Potable Reuse System

The treatment plant operations model was based on detailed data for the Bundamba treatment plant in the Western Corridor Recycled Water Scheme (Lane *et al.* 2011), with the power use models revised to reflect more recent published data (Poussade *et al.* 2011). The model estimates (power use, chemicals use, fugitive gas emissions, sludge generation) were based on (a) the nutrient concentrations in the hypothetical South Caboolture STP effluent feed, and (b) the previous assumptions used for nutrient concentrations in the product water and brine reject streams (Lane *et al.* 2011).

For the product water pumping to North Pine Dam, power demand was estimated at 404 kWh/ML using a simplified hydraulic model. Pipe length and diameter for this calculation were taken from the pre-feasibility costing undertaken for the TWCMP study (BMT WBM 2012). We assumed that the RO plant brine would be discharged by gravity to the Caboolture River estuary, through the existing STP outfall. Given the high salinity of the RO brine, it is possible that restrictions might be imposed on the discharge location. Long distance pumping of the brine would represent another substantial demand for power use that was not considered here.

This study did not consider any possible implications on long-term average rates of evaporation, fugitive CH_4 emissions or water releases from the North Pine Dam. Essentially, our analysis assumed a steady-state dam operation whereby the increased input of the AWTP product water is matched by an equivalent increase in withdrawals from the dam for mains water treatment (Table 3).

Energy use for treating and distributing the water from the dam was informed by previous data collected for Hall *et al* (2009). Assumptions for WTP recovery rate (98.5%), chemicals usage, sludge disposal and mains network supply leaks (13%) were based on data collected for the equivalent operations in the Gold Coast region (Lane *et al.* 2011).

Table 3: Flow balance for the Indirect Potable Reuse (IPR) scenario.

	losses		product flow
	(% losses)	(ML/d)	(ML/d)
AWTP feed flow			16.9
AWTP production	18%	3.0	13.9
net dam evaporation losses	0%	0.0	13.9
water treatment plant operations	1.5%	0.2	13.7
mains network distribution	13%	1.8	11.9
overall (product water to users)	30%	5.0	11.9

3.4. Displaced Water Supplies

3.4.1. Rainwater Tanks

Rainwater tank operations were modelled using the approach developed for the Caboolture TWCMP analysis (Lane *et al.* 2012b). This was based on the same end-use assumptions discussed in the Class A+ end use balance (Table 1), with one exception. Unlike for the Class A+ scenario, householders were assumed to have no preference for mains water use (over rainwater) for outdoor uses - on the premise that rainwater tank supplies are more likely to be considered a "clean" source of water. It was also assumed that households with a rainwater tank (nominally for supplying the laundry) would not have these connected to outdoor taps, and therefore could not affect the rate of Class A+ use for outdoor uses.

Estimates of specific power use for the different rainwater tank end-uses were based on the most comprehensive data on contemporary systems available in Australia (Tjandraatmadja *et al.* 2012a).

A number of key simplifications were made for the analysis in this report. All rainwater tanks were assumed to be 5 kL in size, regardless of the connected end-uses. Inventories related to tank construction were therefore excluded, as they would not change with the introduction of a household Class A+ water supply. All tanks were assumed to use the same sized pump and controller system, regardless of the connected end-uses, and use an automated switch (bypassing the tank) to provide backup mains water. For those parameters described in this paragraph, the choice of more specific (and complex) assumptions would be very dependent on the nature of specific case studies. Our simplified approaches are therefore used to provide indicative analysis only.

3.4.2. Mains Water Supply

Further investigation is required into how best to model mains water supply offsets in the context of planning for future infrastructure in SEQ (Lane *et al.* 2012b). The conventional practice of benchmarking future infrastructure options against some mix of current water supply technologies does not provide any meaningful indication of what would change as a result of a decision to implement (or not) a wastewater reuse scheme that caters for new housing development. What should be modelled are those supply sources that would alter their production rate, as a result of the change (increase or decrease) in mains water demand that is caused by the decision being made. Unfortunately, the complex nature of the integrated SEQ water supply grid makes it difficult to ascertain what this marginal supply technology would be.

Seawater desalination production is commonly discussed as a variable that would change in response to short term perturbations in the regional water cycle (Stickland *et al.* 2012), and longer term growth in regional demand (Hall *et al.* 2009; QWC 2010). We therefore assumed that mains water offsets delivered by the implementation of a wastewater reuse system would lead to an equivalent net reduction in the production from a seawater desalination plant. A companion report to this study provides further discussion on alternative choices for the marginal mains supply (Lane *et al.* 2012b).

Desalination operations were modelled for the Tugun plant, using the approach developed in a previous study (Lane *et al.* 2011), but updated with more published data on the plant's power use (Poussade *et al.* 2011). The power use estimate assumed that 100% of the desalinated product water is distributed within the Gold Coast region. Export of the water from the Gold Coast to the MBRC region was excluded, on the premise that changes in mains usage in the Caboolture area would simply shift the direction (but not quantity) of bulk water transfers around the SEQ region. Ascertaining the net energy use implications of such a change in bulk transfer operations would require further investigation.

3.5. Other Inventories across the System Life-Cycle

Construction

For the full LCA analysis of the South Caboolture Class A+ reuse system, construction inventories were estimated as follows:

- Materials use inventories for construction of the treatment plant were based on detailed estimates for a water treatment plant utilising sand filtration and membranes (Friedrich 2001). While this is not necessarily a good proxy for the particular unit operations used in our scenario, it was considered the most useful data available for generating an indicative estimate of the construction implications.

- Piping inventories for the Class A+ reticulation network were based on the detailed data collected for the Pimpama-Coomera Class A+ system (Lane *et al.* 2011), adjusted for differences in end use characteristics. Materials and transport inputs for pipeline installation were based on the models created for a previous Australian study (Grant *et al.* 2005).

- Construction of rainwater tanks was modelled using the approach described in a previous study for the Gold Coast (Lane *et al.* 2011).

For the GHG comparison across the three different recycling systems, all construction inventories were excluded.

Fertiliser Offsets

Where included, estimates for fertiliser offsets were based on the concentration of nutrients in the wastewater. The calculation approach is described in more detail in a parallel report associated with this UWSRA project (Lane *et al.* 2012a).

Energy Supply; Materials Supply; Transport Operations

Operational inventories (materials use, energy use, emissions) for power generation were taken from the AusLCI database (AusLCI 2012). For the use of chemicals, materials and transport services, the equivalent 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), but where possible, modified to use Australia-specific inventories for power supply, raw materials production, and transport.

4. IMPACT ASSESSMENT MODELS

Table 4: Summary of impact assessment models used in this report.

Impact Category	As Proxy For…	Approach for this Analysis		Summary
Freshwater Extraction (FWE)	ecosystem damage from disruptions to the hydrological cycle (FWE) and discharge of nutrient and organic matter to waterways (EP)	excluded	none of the scenarios have a direct influence on freshwater extraction	*1*
Eutrophication Potential (EP)		customised model	account for local waterway sensitivities	*2*
Ecotoxicity Potential marine / freshwater / terrestrial (METP;FETP;TETP)	ecosystem damage from discharge of organic and metal pollutants	modified version of USES-LCA v2	impact factors for chlorine were excluded, and impact factors for metals were downgraded, to reflect the substantial uncertainty associated with their application	*Chapter 4.2*
Global Warming Potential (GWP)	ecosystem and human health damage from changes to atmospheric concentrations of GHG (GWP) and stratospheric ozone (ODP)	default ReCiPe model		*1*
Ozone Depletion Potential (ODP)		modified ReCiPe model	updated ODP factors for conventional ozone depleting substances & and an ODP factor for N_2O	*2*
Fossil Fuels Depletion (FFD)	availability of critical resources for use by future generations	default ReCiPe model		*1*
Minerals Depletion (MD)		modified ReCiPe model	incorporating an interim impact factor for phosphate rock resources	*2*
Human Toxicity Potential (HTP)	human health damage caused by the emission of organic chemicals and metals to the environment (HTP); and by the emission of substances that cause smog (POFP) or atmospheric particulate accumulation (PMFP)	modified version of USES-LCA v2	impact factors for chlorine were excluded, and impact factors for metals were downgraded, to reflect the substantial uncertainty associated with their application	*Chapter 4.2*
Photochemical Oxidant Formation Potential (POFP)		default ReCiPe model		*1*
Particulate Matter Formation Potential (PMFP)		default ReCiPe model		*1*

1. see (Lane et al. 2012b) 2. see (Lane et al. 2012a)

4.1. Summary

The LCA impact models used in this research project were developed through a detailed review of: (a) key limitations with the default methods currently in use; (b) the implications of these limitations for urban water systems analysis; and (c) a survey of current methodological research underway internationally.

The majority of the chosen impact models were either taken from, or modified from, the ReCiPe impact assessment methodology (Goedkoop *et al.* 2009). Significant changes were made to some of the default LCA impact models during the course of this research project, while for others a number of important challenges were identified.

This report provides details on the review undertaken for *aquatic Ecotoxicity Potential* and *Human Toxicity Potential*, because of the particular focus on toxicity analysis in parallel research on the South Caboolture AWTP (Reungoat *et al.* 2012). Complementary discussion on *Terrestrial Ecotoxicity Potential* is provided in the companion report focussed on conventional wastewater systems (Lane *et al.* 2012a).

Table 4 summarises those particular impact categories and impact models that were used for the analysis presented in this report, and provides a link to the detailed reviews undertaken for other impact categories. The full suite of LCA impact categories that were considered in this research project are summarised in a companion report to this study (Lane *et al.* 2012b).

4.2. Ecotoxicity Potential and Human Toxicity Potential

Toxicity assessment has been a high profile focus in the development of the LCA methodology, and has been the subject of a substantial amount of scientific input. Unfortunately, a number of different toxicity modelling platforms were developed as LCA evolved. These different models produced inconsistent results, and until recently, this acted as an impediment to the widespread uptake of LCA toxicity analysis. However, recent years have seen a shift towards an international consensus amongst the LCA research community, through the development of the USEtox model (Hauschild *et al.* 2008). It is expected that this will increase the level of attention paid to LCA toxicity analysis by the international policy community.

Unfortunately, USEtox only currently provides models for assessment of *Freshwater Ecotoxicity* and *Human Toxicity*. Other LCA-based toxicity packages also consider the M*arine* and T*errestrial* environments. The latter are particularly relevant for analysis of wastewater systems in the SEQ context, given: (a) the vast majority of wastewaters are discharge into estuarine or coastal waterways; and (b) large portions of recycled wastewater is irrigated onto soils.

We have used the toxicity models from the USES-LCA (version 2) modelling package (van Zelm *et al.* 2009), which is the basis of the models implemented in the ReCiPe method (Goedkoop *et al.* 2009). Of the alternatives to USEtox, USES-LCA provides the most recently updated and comprehensive set of toxicity models available for use in LCA. It has also been shown that the USES-LCA impact factors for *Freshwater* and *Human toxicity* impacts correlate reasonably well with those from the USEtox models (Rosenbaum *et al.* 2008). The developers of both USES-LCA and USEtox have confirmed that conclusions drawn using the ReCiPe/USES-LCA package would be representative of the findings reached if using USEtox (Hauschild 2011; Huijbregts 2011).

The USES-LCA package offers flexibility with regards to a number of key assumptions that are employed. These allow for different choices on aspects such as the physical processes and exposure routes that are included in the analysis, the types of toxic effects that are considered, and the uncertainty thresholds that are used to determine which chemicals are considered.

Of primary relevance to this study, are the choices associated with the toxicity impact factors for emissions of metals to the environment. Previous analysis showed that metals emissions are likely to dominate the LCA toxicity results for urban water systems analysis (Foley 2009; Lane *et al.* 2011). However, the research community has recognised a number of shortcomings in the way that metals are handled in LCA toxicity models (Ligthart *et al.* 2004; Diamond *et al.* 2010). The USES-LCA model can be configured to avoid a number of these shortcomings, and this was done for the analysis undertaken in this report. However, one of the key limitations is that the publicly available versions of USES-LCA, nor USEtox, allow for metals speciation effects to be adequately considered. Recent research has demonstrated that, because of this gap, conventional LCA models tend to overstate the significance of metals-related toxicity, often by orders of magnitude (Gandhi *et al.* 2010; Gandhi *et al.* 2011). This shortcoming needs to be recognised when applying LCA toxicity analysis to systems that involve substantial emissions of metals to the environment.

Another modification made for this study, was to exclude the available impact factors for emissions of chlorine and monochloramine. This is relevant to the analysis of wastewater used for irrigation, but also for ascertaining the toxicity implications that are avoided by reducing wastewater discharge to receiving waters. Our previous research showed that chlorine emissions would feature prominently in the LCA toxicity results of wastewater systems if these impact factors are used (Lane *et al.* 2011). However, there are no examples available where these impact factors have been used in the published literature, and there is some uncertainty about whether or not USES-LCA has been appropriately parameterised for these substances. As we were unable to resolve this issue through the course of this project, the decision to exclude chlorine allowed us to focus on other aspects of the LCA toxicity modelling of relevance to wastewater recycling.

5. RESULTS AND DISCUSSION

5.1. LCA for the South Caboolture WRP

The life cycle impacts are shown in Figure 3, normalised against an estimate of the total life-cycle impacts associated with the entire urban water system for the MBRC area. The model for the overall MBRC water system is described in a companion report (Lane *et al.* 2012b).

For each impact category, the scale of the net result (illustrated with the points in Figure 3) indicates how much the overall MBRC system impacts would change if a new Class A+ system (based on this scenario) were to be introduced. Positive values represent a net increase (worsening) in impacts, while negative values represent a net decrease (improvement) in overall impact. The results are broken down into the impacts (positive values) and credits (negative values) associated with different components of the system life cycle.

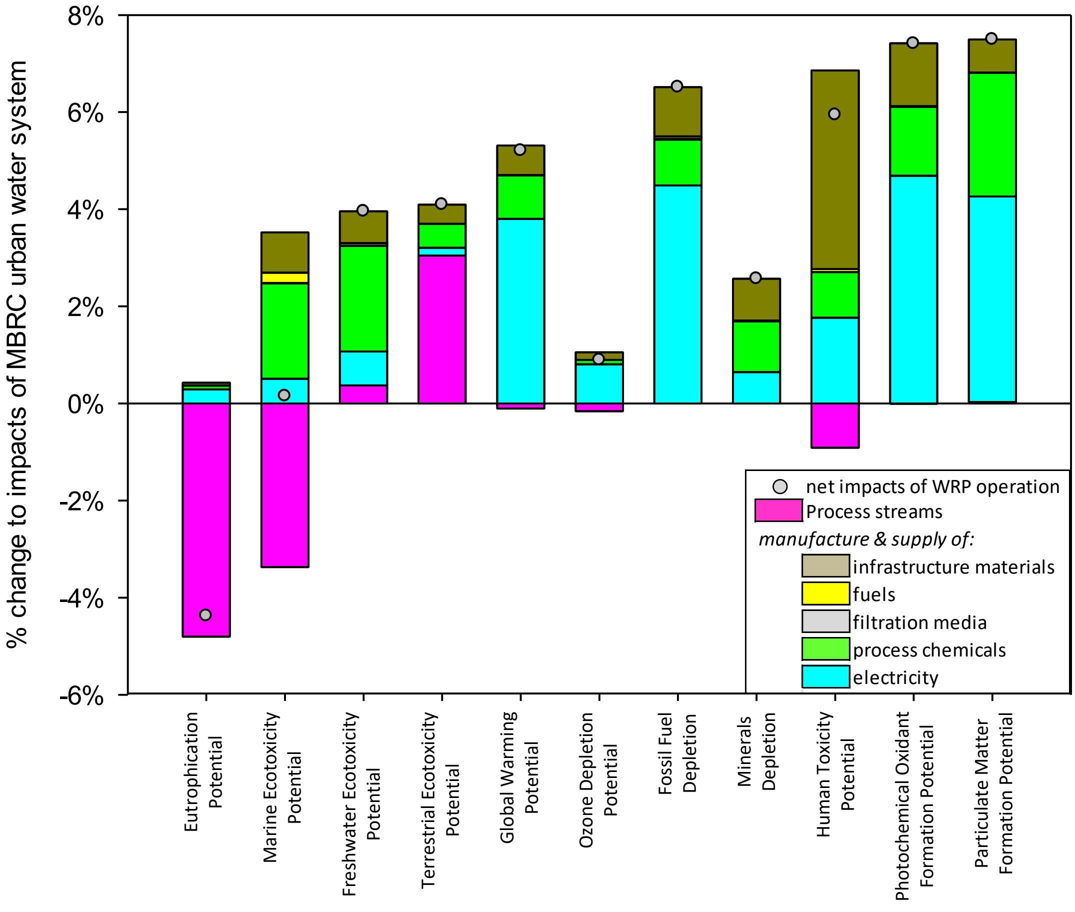

Figure 3: Life cycle impacts associated with a Class A+ reuse scheme supplied by the South Caboolture WRP. Results are expressed as percentage of the total equivalent impacts associated with the entire MBRC urban water system.

The Role of Wastewater Contaminants

The diversion of effluent from estuarine discharge delivers a net reduction in the *Eutrophication Potential* burden of the overall MBRC system. However, the significance of the normalised *Eutrophication Potential* change is small compared to the changes in some other impact categories. This results from our starting-point assumption, that the South Caboolture STP would already be producing effluent with extremely low nutrient concentrations.

The *Eutrophication Potential* associated with power generation is noticeable, but it is thought that the airborne fate models used in LCA substantially overstate the likelihood of power stack NO_X emissions reaching nitrogen-sensitive waterways (Grant *et al.* 2008). Indirect life-cycle *Eutrophication Potential* contributions can therefore be considered as negligible.

Wastewater contaminants also make a substantial contribution to the life-cycle ecotoxicity results, with the diversion from water discharge to irrigation explaining the distinct shift in toxicity burden from the marine environment to the terrestrial environment. The ability to consider the toxicity implications in all these three environmental compartments (Marine, Freshwater and Terrestrial) is one of the distinct advantages with the LCA approach. Unfortunately, there are significant limitations with the available LCA toxicity models that constrain their usefulness for this study. These are discussed in more detail in the following section.

Indirect Impacts across the Life Cycle

All the other impact category results for the South Caboolture treatment plant are dominated by indirect pathways, primarily associated with power generation (Figure 3). The manufacturing and/or supply of chemicals is notable in some impact categories, but are of lesser importance. Similar results were shown by full LCA analysis of other wastewater recycling systems in SEQ (Lane *et al.* 2011). Together, these results could be used to infer that power use is an adequate proxy for much of the indirect environmental burden associated with the lifecycle of wastewater recycling systems.

However care should be taken in extending this conclusion to other recycling systems, because for these same impact categories, other more direct issues could feature under some conditions. For example, wastewater recycling to agriculture could have a substantial effect on P recovery (affecting *Minerals Depletion*), the crop N_2O balance (affecting *Global Warming Potential* and *Ozone Depletion Potential*) and the crop NH_3 balance (affecting *Particulates Formation Potential*). None of these effects would be explained by estimates of power use.

The Challenge with Power Supply Modelling

The dominance of power supply to some of these impact categories might change if different choices were made on the marginal source of power. This is a critical consideration for planning studies, since their recommendations will affect future energy demands over long time frames. Conceptually, the marginal power supply technology is that which would actually change in response to the outcomes (increased or decreased energy demand) of a water utility's planning decision. This might vary depending on the location of the change in demand, whether the change affects peak or offpeak demand, or whether a short or long term timeframe is being considered.

Given the number of different ways in which the marginal power supply could be judged, application of the concept would most likely require that a range of possibilities be considered. Unfortunately, there is relatively little guidance available about how to do so in the Australian context. Further investigation would be required to understand the most appropriate source of marginal power supply for modelling of future urban water systems in SEQ.

Regardless of the methodological approach that might be used, it is important to recognise that the intensity of GHG and other emissions associated with future power use by the water industry might be very different to those associated with the current average supply mix for the Queensland grid. The current practice of using NGERS GHG accounting frameworks in a planning context therefore introduces a potentially significant disconnect between options analysis and the actual implications of planning decisions.

The Significance of Water Supply Offsets

Table 5 presents the same LCA analysis, but this time the results incorporate the reduced need for production of alternative water sources (rainwater tank and mains water supplies).

The offsets are sufficient to negate all but one of the life-cycle environmental impacts associated with the South Caboolture treatment plant operation (Table 5). This finding holds regardless of whether rainwater supplies are assumed sufficient to meet 50% (default) or 100% of the total demand for toilet and external use. The reason that this assumption is not influential is that residential demands make up only 27% of the total Class A+ use in this planning scenario (Table 2).

Table 5: Effect of including water supply offsets, showing normalised results as '*% change to the MBRC urban water system'*. Two sets of results are shown for each impact category – one for the default scenario with a rainwater tank yield of 50%, and another for an extreme case where the tank achieves 100% rainwater yield.

	Eutrophic'n Potential		Marine Ecotox		Freshwater Ecotox		Terrestrial Ecotox		Global Warming		Ozone Depletion		Fossil Fuel Depletion		Minerals Depletion		Human Toxicity		Photochem oxidants		Particulates formation	
AWTP total	-4.4		0.1		4.0		4.1		5.2		0.9		6.5		2.6		5.9		7.4		7.5	
displaced mains supply	-0.7	-0.7	-1.3	-1.3	-1.7	-1.7	-0.4	-0.4	-9.3	-9.3	-2.0	-2.0	-11.0	-11.0	-1.6	-1.6	-4.3	-4.3	-11.5	-11.5	-10.4	-10.4
displaced tanks supply	0.2	0.4	-0.1	-0.3	-0.2	-0.3	0.0	-0.1	-0.9	-1.9	-0.3	-0.5	-1.1	-2.2	-0.2	-0.3	-0.4	-0.9	-1.2	-2.3	-1.1	-2.1
net total	-4.9	-4.7	-1.3	-1.4	2.1	1.9	3.7	3.6	-5.0	-6.0	-1.3	-1.6	-5.6	-6.7	0.8	0.7	1.2	0.7	-5.2	-6.4	-3.9	-5.0
scenario (rain tank yield)	50%	100%	50%	100%	50%	100%	50%	100%	50%	100%	50%	100%	50%	100%	50%	100%	50%	100%	50%	100%	50%	100%

While these effects are very much a function of the chosen scenarios used in this report, the results do illustrate the importance of considering the effect that water supply displacement will have on the overall energy balance. The impact offsets would be lesser if the marginal mains supply was not seawater desalination, or if the Class A+ water were to predominantly displace the use of rainwater tanks. Nonetheless, the results presented in Table 5 suggest that there will be few environmental downsides (from a life-cycle perspective) to the adoption of wastewater recycling, if the scheme can effectively reduce the need for production of alternative water supplies.

5.2. LCA Toxicity Modelling

This section provides a more in depth discussion of the LCA toxicity modelling results. The goal is to provide a link to the detailed chemical and bioanalytical evaluations undertaken in a parallel study on the South Caboolture AWTP (Reungoat *et al.* 2012), and on other aspects of the SEQ urban water system (Macova *et al.* 2010; Poulsen *et al.* 2011).

The Benefits of LCA Toxicity Analysis

The indirect toxicity contributions are predominantly associated with emissions of metals to waterways (*Marine* and *Freshwater Ecotoxicity Potential*) or airsheds from the supply and combustion of fossil fuels (*Terrestrial Ecotoxicity Potential*, *Human Toxicity Potential*). In the case of the *Marine Ecotoxicity* results, the indirect effects are substantial enough to offset much of the benefits associated with eliminating the estuarine discharge of wastewater.

The direct contributions to the *Ecotoxicity Potential* results, associated with discharge or irrigation of wastewater from the South Caboolture treatment system, are also dominated by the very small residual concentrations of metals in the wastewater effluents. This finding is in stark contrast to the focus on organic micropollutants that dominates debate and research effort in SEQ. Notwithstanding the limitations of metals toxicity analysis described in Section 4.2, the dominance of the metals to the overall results can be ascribed to their persistence in the environment. Organic micropollutants, in contrast, will be converted to benign forms in relatively short time frames.

This analysis illustrates how the LCA approach to toxicity modelling could provide a useful, bigger picture perspective on the local concerns about organic micropollutants in wastewater. The potential benefits are threefold.

- Firstly, the LCA modelling framework can identify the potential ecosystem impacts associated with metals emissions even at very low concentrations, thereby providing some insight on whether or not the priority focus on organics is justified.
- Secondly, it allows direct comparisons to be drawn between local, wastewater-related toxicity hazards, and those being caused by electricity generation or transport operations. This could, for example, help to illustrate the true toxicity-related implications of a decision to implement treatment technologies specifically designed to remove organic micropollutants, at the expense of increased system power use.
- Finally, as demonstrated in the previous section (Figure 3) the nesting of the toxicity analysis in the broader LCA framework also means that the toxicity concerns can be considered alongside many other environmental implications (e.g. GHG emissions).

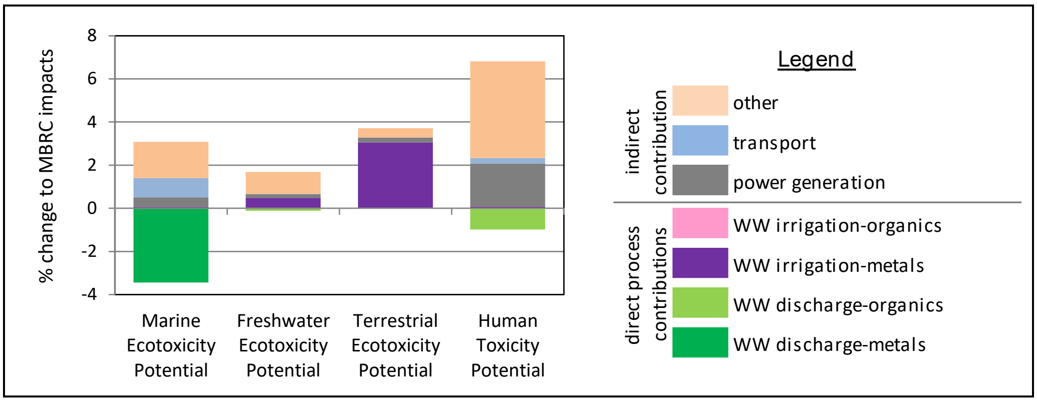

Figure 4: Life cycle ecotoxicity and human toxicity impacts associated with a Class A+ reuse scheme supplied by the South Caboolture AWTP. Results are expressed as percentage of the total equivalent impacts associated with the entire MBRC urban water system. Of the wastewater contaminants, metals make the dominant contribution to the toxicity results, with only a negligible contribution from organic micropollutants.

The Limitations of LCA Toxicity Analysis

Unfortunately, there are a number of limitations with the available LCA toxicity models that prevent adequate conclusions being drawn on life-cycle toxicity related matters in this study.

- As discussed in chapter 4.2, there are major limitations with the way that LCA toxicity models deal with metals emissions. Recent research has shown that improvements to the LCA toxicity modelling approach for metals could reduce the scale of the hazard assessment by an order of magnitude or more for some substances (Gandhi *et al.* 2010; Gandhi *et al.* 2011).

- The coverage of wastewater-related organic chemicals-of-concern in the available LCA toxicity models is poor. We were able to access toxicity impact factors for only 35% of the 54 organic species detected in the South Caboolture treatment plant influent.

- The airshed fate and exposure modelling used in the default LCA toxicity models is based on European conditions, which are not representative of the climatic conditions or population densities found in Australia (Huijbregts *et al.* 2003).

- The scope of the LCA models excludes direct exposure pathways that might be relevant for human toxicity analysis, considering there is the possibility that skin contact or inhalation of Class A+ water could occur in urban settings.

Similarly, the LCA toxicity analysis of the entire urban water system does not consider the direct exposure hazards associated with disinfection byproducts in the mains water system (e.g. Knight *et al.* 2010), nor contaminants in residential rainwater tanks (Huston *et al.* 2012). Such information would be useful if considering the significance (from a broader perspective) of the hazards associated with the wastewater contaminants.

Ongoing work from this project aims to make a number of improvements to the available LCA toxicity models, addressing some of those limitations highlighted above. However, it is important to stress that improved LCA toxicity models would never replace the need for detailed local toxicity assessment and risk analysis. Instead, the goal of this ongoing work is to generate a more robust framework for LCA toxicity analysis of urban water systems that could complement the more locally focussed approaches.

Comparison of Different Approaches

There is an additional scope of work that could also prove fruitful. This would specifically investigate how LCA toxicity assessment could be directly benchmarked against, and even utilise, the chemical and bio-analytical toxicity methodologies being used and developed through other SEQ-based research on wastewater pollutants (Poulsen *et al.* 2011).

To an extent, the LCA toxicity models are similar to the additive TEQ approach used by Reungoat *et al* (2012). Both approaches combine emission loads with relative toxicity factors taken from species sensitivity distributions (using EC50 values), and sum the total toxicity across a number of chemicals. However the TEQ analysis was based on EC50 values for a single bacterial species, in order to maintain consistency with the bioanalytical microtox assay (Reungoat *et al.* 2012). In contrast, the convention in LCA is to incorporate the EC50 values of a range of species and a range of genera, in order to provide a more holistic assessment of the impacts that might occur.

A further difference between the LCA and TEQ approaches is that LCA extends the analysis to include long term fate and exposure models for the wastewater pollutants in the receiving aquatic environment. This is analogous to the fate and exposure modelling undertaken in environmental risk assessment. However, there are fundamental differences between risk assessment and conventional approaches to toxicity modelling for LCA (de Haas *et al.* 2006). A primary example is that LCA includes consideration of all marginal releases, regardless of their scale or the presence of any local threshold effects. This need not necessarily be a constraint in LCA, as previous studies have demonstrated how local ambient conditions could be directly considered in LCA toxicity modelling (Sleeswijk *et al.* 2010).

The most substantial benefits might come from enabling LCA toxicity models to incorporate the insight developed from the extensive use of bioassay analysis in SEQ. For example, analysis on wastewater streams at the South Caboolture site illustrated that chemical-based toxicity assessment (the TEQ approach) might account for only a small fraction of the toxicity detected by the more comprehensive bioassay based methods (Reungoat *et al.* 2012). Since the LCA approach also relies on an additive, chemical-based approach, the implications of this finding might be equally relevant.

Utilising information from predictive mixture toxicity models and experimental bioanalytical toxicity data would help to address some of the gaps noted above. This would improve the quality of LCA based analysis on organics vs. metals-related toxicity, and more generally, for comparing the toxicity contributions across different parts of the system life cycle. The SEQ research community would seem well placed to investigate this opportunity, given it has access to an unprecedented level of data and expertise in chemical and bioassay analysis for urban water systems.

5.3. Comparison of Different Recycling Systems

Power Use for Wastewater Treatment

Power use for the AWTP treatment process is the biggest contributor to the overall GHG footprints of each system (Table 6). The ozone-BAC treatment process requirements are surprisingly close to that needed for advanced wastewater treatment using RO membranes and advanced oxidation, and far greater than the equivalent for the ultrafiltration (UF) plant.

One reason for the high power demand for ozone-BAC treatment is the high backwash rate that we adopted. The product water recovery for the ozone-BAC model (83%) is similar to that for the RO based process (82%), and much lower than that assumed for the UF-based process (98%). The effect of these differences can be seen most clearly in a comparison of the power use allocated to the effluent feed pumps. Even though the actual power consumption is identical across all three scenarios, the higher recovery for the UF process translates to lower effective power intensity (113 kg-CO2e/ML vs. ~135 kg-CO2e/ML) when compared on a product water basis.

The power use models for the treatment steps are, however, the main cause of the treatment-related variation shown in Table 6. Unfortunately, both the South Caboolture and Pimpama treatment plants collect only a single power use reading that incorporates the entire treatment process and the product water distribution pumps. Therefore, it was not possible to ascertain the cause of the differences shown between these two processes.

Table 6: Greenhouse gas (GHG) footprints of three different wastewater recycling systems

		GHG footprint (kg-CO2e) per ML of product water generated[Δ]			GHG footprint (kg-CO2e) per ML of mains water offsets		
		direct non-potable reuse		indirect potable reuse	direct non-potable reuse		indirect potable reuse
		Type 1 (Ozone-BAC)	Type 2 (UF)	(RO-AOP)	Type 1 (Ozone-BAC)	Type 2 (UF)	(RO-AOP)
power use	AWTP feed pumps*	136	113	135	264	219	158
	AWTP treatment	703	257	1067	1369	500	1247
	AWTP product water pumping	252	252	371	490	490	433
	mains water system^	--	--	--	--	--	259
	rainwater tanks operation^^	--	--	--	-661	-661	--
other	fugitive gas emissions	-3	-3	-7	0	0	-8
	chemicals use	269	74	123	524	145	322
	media use	1	3	4	2	6	5
	misc	2	1	4	3	2	4
	total	1360	696	1697	1992	1020	2420

[Δ] taken at the point of discharge from the treatment plant

** assumed to be 108 kWh/ML of feed water processed by each of the AWTP technologies*

^ includes treatment and distribution through the mains water network

^^ rainwater tanks operation displaced by provision of Class A+ water

Our estimate for power consumption (703 kWh/ML of feed) in the treatment train of the South Caboolture plant is based on detailed production data, however is more than double the ~27 0kWh/ML used in two other studies on ozonation-based technology for wastewater treatment (Schimoller *et al.* 2008; Munoz *et al.* 2009). Further review is required to determine whether the South Caboolture AWTP power use reading is being influenced by peripheral demands (e.g. power use for buildings and services). If not, then the available data suggests that there might be some potential for power use efficiency improvements at the South Caboolture plant. A more detailed power use audit for the treatment plant might therefore be beneficial.

Class A+ Water Distribution

The results in Table 6 suggest that product water distribution is likely to be a significant component of the GHG footprint for Class A+ recycling systems. However, better quality power use estimates for Class A+ product water pumping would be required to ascertain the true significance of this component. Our assumptions for Class A+ pumping were based on a mixture of interpolation and engineering estimates across the different end use types in the Caboolture reuse scenario. Using these, the product water distribution represented 19% and 36% of the total GHG footprint for the ozone-BAC and UF based systems respectively. The weak point of our estimates was the lack of empirical data for distribution to the complex network of household and commercial users of Class A+ water.

Neither of the two major Class A+ systems in SEQ collate power use data specifically for the product water pumping, and this issue should be given a greater focus in the modelling undertaken for water system planning studies. Planning estimates would benefit from high quality empirical data being available, but might also require detailed hydraulic modelling, given that the power demand could vary substantially depending on location specific factors. Our results demonstrate how significant the pumping estimates could be to GHG comparisons of alternative water supply technologies.

Chemicals Used for Wastewater Treatment

Chemicals use is the other significant contributor to the GHG footprints of the three treatment systems, primarily associated with the chemicals manufacturing, rather than the product transfer to the treatment plant site. While these results are based on real chemicals usage data, the quality of the 'embedded' GHG estimates is uncertain because of the reliance on database information for the manufacturing processes. An equivalent assessment of the Bundamba RO-AOP treatment plant (Poussade *et al.* 2011), but using a different database for chemical manufacturing, found the chemicals-related GHG footprint to be less significant to the total than in our study.

Nonetheless, the results highlight that energy usage alone is not a sufficient basis for undertaking GHG based comparisons of wastewater recycling schemes and alternative water supply approaches. Furthermore, the relatively high chemicals usage of some wastewater related technologies could represent a point of unwanted exposure to cost increases as a result of carbon pricing schemes being adopted in Australia. As an example, Sydney Water have estimated that their costs for carbon-intensive supply-chain items could increase by greater than $5 million/yr as a result of suppliers passing on their carbon price liability (Sydney Water 2011).

The Importance of Considering Mains Water Displacement

When the systems are compared on the basis of product water generated (using the *cradle-to-gate* approach), the South Caboolture Class A+ reuse scenario has a 20% lower GHG footprint than the IPR system. However, the effective delivery from product water to mains water savings is much lower for the Class A+ system. Only 55% of the Class A+ product actually displaces mains water use in our scenario (Table 2), whereas the equivalent ratio for the IPR system is 70% (Table 3). As a result, when normalised on the basis of their capacity to offset mains water, the GHG intensity of the two treatment systems is approximately equal. It is only the collateral credit for avoiding rainwater tank operations that provides a point of difference.

This illustrates the downside of Class A+ reuse systems becoming overly dependent on irrigation for disposal of the treated wastewater. Our results clearly indicate that it would be preferable to direct the Class A+ water to higher value users that would otherwise be dependent on mains supply.

By connecting directly into the mains supply grid, IPR systems avoid completely the challenges of supply-demand imbalances and user selection. If it could be demonstrated that ozone-BAC treatment is a suitable substitute for RO membranes, then this would deliver an IPR system with a much lower GHG footprint than conventional RO-based approaches.

5.4. End-Use and Offsets Analysis – Challenges and Implications

The Potential for Financial and Environmental Disconnects

Our analysis has demonstrated the sensitivity of environmental outcomes to the amount of mains water offsets that can be delivered by wastewater recycling systems (Table 5).

This highlights the need for planning studies to base their analysis of Class A+ wastewater recycling on realistic end use assumptions. Once implemented, direct reuse systems are constrained by the number of connected users and the specific end-use types that are allowed. The implications of using poorly informed end-use assumptions could therefore be a significant and unexpected shortfall in reuse demand. While knowledge of SEQ end use behaviour has greatly improved over recent years, there remain considerable uncertainties about how water usage will change over time.

One issue that has received scant attention is the potential for householders to bias against Class A+ use in the event that they have an alternative for outdoor water use. The default assumption in our scenarios, following the approach used in the MBRC planning study, was that households connected with Class A+ supply would also have an onsite rainwater tank connected to laundry uses. It seems plausible that many such households would choose to also connect the rainwater tank to at least one external tap. In a previous Gold Coast study, householders stated a preference to use rainwater, rather than Class A+ recycled water, for certain outdoor tasks (Willis *et al.* 2009). While this survey was undertaken prior to the commissioning of the Class A+ supplies, it highlights the potential for conflict between these two alternative supply sources. Follow up studies might be warranted to determine how these interactions play out in practice.

Fundamental shortfalls in demand for product water might introduce significant cost pressures for Class A+ scheme operators. One approach to addressing such a shortfall would be to expand the reuse network, an outcome that is clearly supported by our environmental impact-based analysis. However, the water utility responsible for the Class A+ scheme would have to meet the full capital and operating costs associated with such a step. If they do not realise any direct financial savings from the resulting water supply benefits (reduced rainwater tank use and/or reduced mains water supply from the SEQ grid), then this might not seem a particularly attractive choice.

An alternate approach might be to encourage additional wastewater reuse (e.g. a higher demand per household) by the existing users. While this would increase the AWTP operator's ability to recover costs incurred for the existing infrastructure, it could lead to perverse environmental outcomes. Additional volumetric use for the same end-use needs does not deliver additional offsets of alternative water supplies, and therefore does not generate an increase in the associated environmental benefits. Furthermore, an increase in AWTP throughput will come at the expense of increased power and chemical inputs that are not insubstantial.

From an environmental perspective, this could only be justified if the life-cycle implications of increased operation are considered less important than the benefits associated with diverting additional wastewater from estuarine discharge. The lower the nutrient concentrations in the STP effluent (AWTP feed water), the harder it will be to reach this conclusion. Furthermore, the lower the secondary effluent nutrient concentrations, the more likely it is that there are other, more cost-effective, options available for reducing total nutrient inputs to Moreton Bay.

Modelling the Offsets

Even if the quantity of water supply offsets can be determined, there remain considerable challenges for estimating the likely implications of avoided water production.

For residential systems that would otherwise be using rainwater tank supplies, meaningful predictions of how much rainwater and mains water might be displaced can only be resolved with high quality, locally specific tank yield modelling. Recent UWSRA research has highlighted that conventional approaches to tank yield modelling might substantially overestimate the effectiveness of this approach to residential water supply (Coultas *et al.* 2012; Lane *et al.* 2012b).

Furthermore, there is the challenge of using realistic energy use assumptions for rainwater tank supplies. While recent research has greatly improved our understanding on the energy burden of conventional rainwater tank systems, it has also highlighted the potential for rainwater tank supplies to be delivered at a much lower energy intensity than currently seems to be the case (WCG 2009; Tjandraatmadja *et al.* 2012b). For developments where rainwater tanks will be installed progressively over a number of years, benchmarking Class A+ supplies against data on current tank performance could therefore be misleading.

Estimating the impacts associated with displaced mains supply will be equally problematic, given uncertainty about how best to determine the marginal supply technology of relevance to any particular planning decision. This issue is discussed in more depth in a companion report from this project (Lane *et al.* 2012b).

6. CONCLUSIONS

The Life-Cycle Environmental Implications of Class A+ Wastewater Recycling from the South Caboolture AWTP

The ozone-BAC technology in use at the South Caboolture Water Reclamation Plant can produce product water with a lower GHG footprint than a reverse osmosis based process. Given encouraging findings about the capacity for this technology to remove organic micropollutants in urban wastewater, the ozone-BAC combination warrants inclusion in any future investigations into indirect potable reuse in Australia.

However, power consumption at the ozone-BAC plant appears substantially higher than the few published benchmarks available. It may be that there are opportunities for process efficiency improvements at the South Caboolture plant. Along with high treatment power use, the high rate of backwash filtration control represents a substantial loss of product water capacity.

Power consumption at the South Caboolture plant also appears substantially higher than for a Class A+ production plant utilising UF treatment. Unfortunately, analysis of both technologies is constrained by a reliance on highly aggregated data. One particular weakness is the lack of attention paid to the energy demand for reticulation of Class A+ water. This could represent a substantial portion of the GHG footprint of direct reuse systems, yet our observation is that this item frequently gets overlooked in planning studies or comparative analysis.

Furthermore, comparisons across different recycling technologies should be based on more than just differences in electricity use. Chemicals use, and potentially even fugitive emissions, can play a key part in the GHG footprints of wastewater recycling systems. It is also not a given that GHG analysis will provide a good proxy for the other environmental externalities that occur across the system life cycle. A more comprehensive comparison, across a broader set of life-cycle environmental indicators, would likely reveal some important tradeoffs.

Despite the apparently high electricity use at the South Caboolture plant, it is clear that Class A+ reuse systems could deliver net environmental benefits if the need for mains water supplies can be reduced.

However, if they are to consider Class A+ systems, the key challenge for decision makers will be to base their planning on: (1) realistic demand profiles; (2) realistic assumptions on how much this demand will actually displace some alternative source; and (3) realistic assumptions on what alternative source will actually be displaced. Not doing so is likely to introduce a future disconnect between the financial imperatives of the water utility, and the environmental benefits that the Class A+ scheme could deliver.

A Comparison of Different Treatment Technologies and Reuse Systems Relevant to Urban Water Planning in SEQ

Comparison across the Class A+ and IPR reuse approaches makes it clear that direct, non-potable reuse could offer substantial benefits, if it facilitates treatment processes with a low GHG footprint and low energy pumping. But it must also match the inherent capacity of the IPR concept to displace mains water supplies. A dependence on large-scale irrigation of urban spaces may not deliver such benefits. Direct reuse in industrial facilities might be an attractive candidate for generating Class A+ demand, however, the potential for ozone-BAC and other low energy technologies to deliver suitable quality water would need further investigation.

Modelling the displacement of alternative water supplies becomes more complex when households are involved. In the Queensland context, where rainwater tanks are the norm for new development and are likely to coexist with Class A+ schemes, high quality analysis of rainwater tank performance (yield and energy demands) will be just as important as high quality analysis of advanced wastewater treatment systems. Furthermore, residential demands for Class A+ water will be susceptible to householder preferences on the day to day choice between mains water, rainwater tank, and Class A+ water supplies. Addressing this major knowledge gap could be critical if residential-based Class A+ reuse schemes are to play a significant role in the future of SEQ's urban water system.

Benefits and Constraints with Using LCA for Analysis of Wastewater Recycling Systems

Life-cycle thinking could improve the analysis of urban wastewater recycling options in a number of other ways.

Assumptions made on the marginal supply sources of water supply, and power supply, will be critical to any conclusions that are drawn from such studies. In both cases, there is a clear need to move beyond a reliance on status quo information, so that planners recognise the true implications of decisions made on long term infrastructure. To achieve this, however, decision makers will require structured guidance on the most appropriate quantitative assumptions for use in the planning context.

If implemented well, the life-cycle approach to toxicity analysis could provide important perspectives for decision making. Robust LCA toxicity analysis, when combined with locally specific monitoring and risk assessment, would highlight whether efforts to reduce local toxicological risks are shifting these to elsewhere in the system life cycle, or creating unacceptably large tradeoffs with respect to other environmental issues.

Unfortunately, the current status of LCA toxicity models provides a significant barrier to such an undertaking, as they lack the necessary qualities to provide meaningful interpretation of wastewater systems performance in Australia. Further research effort could therefore deliver substantial improvements in the capacity for SEQ decision makers to assess the true environmental implications of wastewater recycling. A key challenge will be to combine the insights from LCA toxicity analysis with recent advances in mixture-based toxicity assessment.

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
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

REFERENCES

AusLCI (2012) Australian Life Cycle Inventory Database (AusLCI). Australian Life Cycle Assessment Society.

Bauman H, Tillman A (2004) 'The hitch hiker's guide to LCA: an orientation in life cycle assessment methodology and application.' (Studentlitteratur: Lund, Sweden).

Beal, C.D. and Stewart, R.A. (2011). South East Queensland Residential End Use Study: Final Report. Urban Water Security Research Alliance Technical Report No. 47.

Coultas EH, Maheepala S, E. NL, C. B, Stewart R, Chong MN, Sharma AK (2012) Towards the Quantification of Rainwater Tank Yield in South East Queensland by Considering the Spatial Variability of Tank Supplies, Brisbane.

de Haas D, Foley J, Barr K (2008) Greenhouse gas inventories from WWTPs - the trade-off with nutrient removal. Water Environment Federation, National Harbour, Maryland, USA.

de Haas D, Foley J, Lant P (2009) Energy and greenhouse footprints of wastewater treatment plants in South-East Queensland. Australian Water Association, Melbourne.

de Haes HAU, Sleeswijk AW, Heijungs R (2006) Similarities, differences and synergisms between HERA and LCA - An analysis at three levels. Human and Ecological Risk Assessment 12, 431-449.

Diamond ML, Gandhi N, Adams WJ, Atherton J, Bhavsar SP, Bulle C, Campbell PGC, Dubreuil A, Fairbrother A, Farley K, Green A, Guinee J, Hauschild MZ, Huijbregts MAJ, Humbert S, Jensen KS, Jolliet O, Margni M, McGeer JC, Peijnenburg W, Rosenbaum R, van de Meent D, Vijver MG (2010) The clearwater consensus: the estimation of metal hazard in fresh water. International Journal of Life Cycle Assessment 15, 143-147.

Foley J (2009) Life Cycle Assessment of Wastewater Treatment Systems. University of Queensland.

Friedrich E (2001) Environmental Life Cycle Assessment of Potable Water Production. University of Natal.

Frischknecht R, Jungbluth N, Althaus H-J, Doka G, Dones R, Hischier R, Hellweg S, Nemecek T, Rebitzer G, Spielmann M (2007) 'Overview and Methodology. Final report ecoinvent data v2.0, No. 1.' Swiss Centre for Life Cycle Inventories, Dübendorf, CH.

Gandhi N, Diamond ML, Huijbregts MAJ, Guinee JB, Peijnenburg W, van de Meent D (2011) Implications of considering metal bioavailability in estimates of freshwater ecotoxicity: examination of two case studies. International Journal of Life Cycle Assessment 16, 774-787.

Gandhi N, Diamond ML, van de Meent D, Huijbregts MAJ, Peijnenburg W, Guinee J (2010) New Method for Calculating Comparative Toxicity Potential of Cationic Metals in Freshwater: Application to Copper, Nickel, and Zinc. Environmental Science & Technology 44, 5195-5201.

Goedkoop M, Heijungs R, Huijbregts MAJ, De Schryver A, Struijs J, van Zelm R (2009) 'ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level.' Pre Consultants, CML University of Leiden, Radboud University, RIVM Bilthoven, Netherlands.

Grant T (2012) Simapro Australasian Database v2012.6. Life Cycle Strategies.

Grant T, Opray L (2005) 'LCA report for sustainability of alternative water and sewerage servicing options.' RMIT Centre for Design, Melbourne.

Grant T, Peters GM (2008) 'Best Practice Guide to Life Cycle Impact Assessment in Australia.' AusLCI & ALCAS.

Hall M, West J, Lane J, de Haas D, Sherman B (2009) 'Energy and Greenhouse Gas Emissions for the SEQ Water Strategy.' Urban Water Security Research Alliance, Brisbane.

Hauschild MZ (2011) personal communications.

Hauschild MZ, Huijbregts M, Jolliet O, MacLeod M, Margni M, van de Meent DV, Rosenbaum RK, McKone TE (2008) Building a model based on scientific consensus for life cycle impact assessment of chemicals: The search for harmony and parsimony. Environmental Science & Technology 42, 7032-7037.

Huijbregts MAJ (2011) personal communications.

Huijbregts MAJ, Lundi S, McKone TE, van de Meent D (2003) Geographical scenario uncertainty in generic fate and exposure factors of toxic pollutants for life-cycle impact assessment. Chemosphere 51, 501-508.

Huston R, Chan YC, Chapman H, Gardner T, Shaw G (2012) Source apportionment of heavy metals and ionic contaminants in rainwater tanks in a subtropical urban area in Australia. Water Research 46, 1121-1132.

Knight N, Shaw G, Sadler R, Wickramasinghe W (2010) 'Disinfection By-product Formation and Minimisation in South East Queensland Drinking Water.' Urban Water Security Research Alliance, Technical Report No. 23, Brisbane.

Lane J, de Haas D, Lant P (2011) 'Life cycle assessment of the Gold Coast urban water system.' Urban Water Security Research Alliance, Technical Report No. 52, Brisbane.

Lane J, Lant P (2012a) 'Application of Life Cycle Assessment to wastewater systems planning.' Urban Water Security Research Alliance, Technical Report No. 97, Brisbane.

Lane J, Lant P (2012b) 'Using Life Cycle Assessment to inform Total Water Cycle Management Planning - an investigation for the Caboolture catchment.' Urban Water Security Research Alliance, Technical Report No. 85, Brisbane.

Lee CO, Howe KJ, Thomson BM (2012) Ozone and biofiltration as an alternative to reverse osmosis for removing PPCPs and micropollutants from treated wastewater. Water Research 46, 1005-1014.

Ligthart T, Aboussouan L, van de Meent D, Schönnenbeck M, Hauschild M, Delbeke K, Struijs J, Russell A, Udo de Haes H, Atherton J, van Tilborg W, Karman C, Korenromp R, Sap G, Baukloh A, Dubreuil A, Adams W, Heijungs R, Jolliet O, de Koning A, Chapman P, Verdonck F, van der Loos R, Eikelboom R, Kuyper J (2004) Declaration of Apeldoorn on LCIA of Non-Ferrous Metals. http://www.leidenuniv.nl/cml/ssp/projects/declaration_of_apeldoorn.pdf.

Macova M, Escher BI, Mueller JF, Toze S (2010) 'Bioanalytical tools to evaluate micropollutants across the Seven Barriers of the Indirect Potable Reuse Scheme.' Urban Water Security Research Alliance, Technical Report No. 30, Brisbane.

Munoz I, Rodriguez A, Rosal R, Fernandez-Alba AR (2009) Life Cycle Assessment of urban wastewater reuse with ozonation as tertiary treatment A focus on toxicity-related impacts. Science of the Total Environment 407, 1245-1256.

NHMRC & NRMMC (2011) 'Australian Drinking Water Guidelines ' National Health and Medical Research Council, National Resource Management Ministerial Council, Canberra.

Poulsen A, Chapman H, Leusch F, Escher B (2011) 'Application of Bioanalytical Tools for Water Quality Assessment.' Urban Water Security Research Alliance, Technical Report No. 41, Brisbane.

Poussade Y, Vince F, Robillot C (2011) Energy consumption and greenhouse gases emissions from the use of alternative water sources in South East Queensland. Water Science and Technology: Water Supply 11, 281-287.

QG (2008a) Queensland Development Code part MP 4.2 - Water Savings Targets. http://www.dip.qld.gov.au/building/current-parts.html.

QG (2008b) Water quality guidelines for recycled water schemes, Brisbane. http://www.derm.qld.gov.au/water/regulation/recycling/guidelines.html.

QWC (2010) 'South East Queensland Water Strategy.' Brisbane.

Reungoat J, Escher BI, Macova M, Farre MJ, Argaud FX, Rattier M, Gernjak W, Keller J (2012) 'Wastewater Reclamation using Ozonation combined with Biological Activated Carbon Filtration.' Urban Water Security Research Alliance, Technical Report No. 69, Brisbane.

Rosenbaum RK, Bachmann TM, Gold LS, Huijbregts MAJ, Jolliet O, Juraske R, Koehler A, Larsen HF, MacLeod M, Margni M, McKone TE, Payet J, Schuhmacher M, van de Meent D, Hauschild MZ (2008) USEtox-the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. International Journal of Life Cycle Assessment 13, 532-546.

Schimoller L, Bellamy B, Curl J, Helmy M (2008) Alternative Treatment for Indirect Potable Reuse. WateReuseAssociation, Dallas, Texas, USA.

Schnoor JL (2009) LCA and Environmental Intelligence? Environmental Science & Technology 43, 2997-2997.

Sleeswijk AW, Heijungs R (2010) GLOBOX: A spatially differentiated global fate, intake and effect model for toxicity assessment in LCA. Science of the Total Environment 408, 2817-2832.

Stewart R (2011) 'Verifying potable water savings by end use for contemporary residential water supply schemes.' Waterlines Report Series No 61, Canberra.

Stickland S, Moffatt J, Stevenson S (2012) Water supply resilience - A grid for all seasons. Australian Water Association, Sydney.

Sydney Water (2011) 'Sydney Water submission to IPART - Review of prices for Sydney Water Corporation's water, sewerage, stormwater and other services.' Sydney Water.

Tjandraatmadja G, Pollard C, Sharma AK, Gardner T (2012a) Dissecting Rainwater Pump Energy Use in Urban Households. Urban Water Security Research Alliance, Brisbane, Queensland.

Tjandraatmadja G, Pollard C, Sharma AK, Gardner T (2012b) 'Optimisation of Energy Use in Household Rainwater Supply Systems.' Urban Water Security Research Alliance, Technical Report No. 89, Brisbane.

van Zelm R, Huijbregts MAJ, van de Meent D (2009) USES-LCA 2.0-a global nested multi-media fate, exposure, and effects model. International Journal of Life Cycle Assessment 14, 282-284.

WBM (2012) 'Total Water Cycle Management Plan for MBRC.'

WCG (2009) 'Energy Consumption in Domestic Rainwater Harvesting.' Water Conservation Group, Heatherton, Victoria.

Willis R, Stewart RA, Emmonds S (2009) Pimpama-Coomera dual reticulation end use study: baseline situational context and post commission end use prediction, Brisbane.

Willis RM, Stewart RA, Williams PR, Hacker CH, Emmonds SC, Capati G (2011) Residential potable and recycled water end uses in a dual reticulated supply system. Desalination 272, 201-211.

Urban Water Security Research Alliance