Australian Water Recycling
Centre of Excellence

Sustainability Literature Survey (Sub-stream 1.4)

Report for the National Demonstration, Education and Engagement Program

Report for the National Demonstration, Education and Engagement Program

This report has been prepared as part of the National Demonstration Education and Engagement Program (NDEEP). This Program has developed a suite of high quality, evidence-based information, tools and engagement strategies that can be used by the water industry when considering water recycling for drinking purposes. The products are fully integrated and can be used at different phases of project development commencing at "just thinking about water recycling for drinking water purposes as an option" to "nearly implemented".

Project Leader

David de Haas
GHD
145 Ann Street
Brisbane QLD 4000 AUSTRALIA
Telephone: +61 7 3316 3000

Contact: David de Haas   [David.deHaas@ghd.com]

Partners

University of New South Wales
Yarra Valley Water

About the Australian Water Recycling Centre of Excellence

The mission of the Australian Water Recycling Centre of Excellence is to enhance management and use of water recycling through industry partnerships, build capacity and capability within the recycled water industry, and promote water recycling as a socially, environmentally and economically sustainable option for future water security.

The Australian Government has provided $20 million to the Centre through its National Urban Water and Desalination Plan to support applied research and development projects which meet water recycling challenges for Australia's irrigation, urban development, food processing, heavy industry and water utility sectors. This funding has levered an additional $40 million investment from more than 80 private and public organisations, in Australia and overseas.

This Literature *Review ("Report") has been prepared by GHD Pty Ltd ("GHD") for the University of New South Wales and the Australian Water Recycling Centre of Excellence ("UNSW/ AWRCOE").*

GHD and its servants, employees and officers otherwise expressly disclaim responsibility to any person other than UNSW/ AWRCOE arising from or in connection with this Report.

To the maximum extent permitted by law, all implied warranties and conditions in relation to the services provided by GHD and the Report are excluded unless they are expressly stated to apply in this Report.

The services undertaken by GHD in connection with preparing this Report:

- *were limited to those specifically detailed in Section 1 of this Report;*

- *did not include comprehensive coverage of all sustainability issues related to urban water systems.*

The opinions, conclusions and any recommendations in this Report are based on assumptions made by GHD when undertaking services and preparing the Report ("Assumptions"), including (but not limited to):

- *the stated assumptions in Section 1 of this Report;*

- *that information contained in the published literature references surveyed is factually correct;*

- *that information contained in the reports or other material (published or unpublished) referenced and supplied for use in this study by other partners in the above-mentioned AWRCOE program is factually correct; and*

- *by supplying information contained in reports or other material (published or unpublished) to GHD that permission is given by the relevant external parties or project partners to GHD to cite such information for the purposes of this study in this report.*

GHD expressly disclaims responsibility for any error in, or omission from, this Report arising from or in connection with any of the Assumptions being incorrect.

Subject to the paragraphs in this section of the Report, the opinions, conclusions and any recommendations in this Report are based on conditions encountered and information reviewed at the time of preparation and may be relied on for the duration of the AWRCOE NDEEP project (terminating on or before 28 November 2014), after which time, GHD expressly disclaims responsibility for any error in, or omission from, this Report arising from or in connection with those opinions, conclusions and any recommendations.

Contents

Table Index

Figure Index

Executive Summary

It is anticipated that communicating issues around Sustainability will be an important part of the National Demonstration Education and Engagement Program (NDEEP) for this project. This Literature Review addresses the Milestone 1 deliverable under Sub-stream 1.4 (Sustainability) of this project.

Fundamentally, questions around Sustainability of recycled water as an alternative supply of drinking water will need to be framed in the context of a comparison against a 'base case' (for example Business As Usual, or Desalination). Depending on the context of the questions, the driver for adopting recycled water may be to match water supply to either a growth in demand or a shortage of supply from other sources, including conventional ones in the 'base case'. Where growth in demand, relative to supply is the fundamental driver, careful delineation is necessary between the *sustainability of growth itself* versus the *relative sustainability* of one supply option compared to another, by whatever metrics of sustainability applied.

Sustainability ultimately is focussed on the constraints posed by the finite resources of the planet to human activity. The form and implementation of those constraints, including the limits of growth or consumption, requires concerted global effort, as seen for example in the current debate around Climate Change issues. As part of the NDEEP, it is anticipated that this project can effectively address *relative sustainability* by comparing different water supply options. This will help to engage stakeholders in debate around ultimate sustainability limits. However, it is not expected that the NDEEP will be able effectively address underlying questions around growth in water demand, although these do arise in an Australian context, amongst others. This distinction will need to be communicated at the outset in the NDEEP.

In one of the first applications of Life Cycle Assessment (LCA) to an urban water system in Australia, Lundie and co-workers (2004) provided insightful summary points, namely:

1. *"LCA provides a defensible methodological platform on which to quantify environmental burdens associated with the base case and on which alternative future systems can be compared on a quantitative basis. LCA allows environmental benchmarking of 'business as usual' against promising alternatives for sustainable water services.*

2. *Using LCA early on in the planning process helps ensure that environmental issues are considered…. Performing LCA has enabled (the water utility) to capture environmental effects associated with the consumption of materials, which does not routinely occur in other strategic planning processes".*

3. *Building an LCA model on the scale and complexity of an entire large urban water system… is a resource-intensive process.*

4. *The process of constructing the LCI (Life Cycle Inventory) involves information exchanges between planning and operational staff which can enhance communication in a large organization. It has also provided information which has enriched communication with external stakeholders".*

It is a fundamental requirement of LCA methodology that it requires definition of the system boundary for the cases being studied (whether hypothetical or real) and the collection of a large body of inventory data for the included processes within that system. Valid comparison of the LCA outputs for two cases (e,g. alternative vs. base case) requires that the system boundaries be the same, although processes within

them may differ. Hence, when comparing, for example, water recycling with desalination, an equitable comparison of upstream and downstream impacts is necessary for a meaningful LCA. Upstream impacts might include, for example: power generation; chemicals manufacture and transport; and treatment of wastewater as the source of water for recycling, or seawater for desalination). Downstream impacts might include, for example: end product distribution and/or discharge (e.g. recycled water into a freshwater impoundment); and waste stream transport and disposal, including resultant offsets (e.g. biosolids vs. fertiliser use in agriculture).

The next step of this project will be to test the extent to which a case study (hypothetical or real) can be constructed as part of the NDEEP for the purposes here of illustrating the value of LCA in communicating sustainability issues. There are sufficient examples of LCA outputs in the literature to illustrate the likely form and possible content of LCA models in the context of alternative urban water supply options. It is recommended that using the full LCA approach be contrasted against a simpler approach by canvassing opinions across a range of stakeholders (e.g. community, planners, regulators, educators). The simpler approach would be to use selected environmental sustainability indicators, based on literature data, to compare alternative urban water supply options.

Examples of environmental sustainability indicators for which data is likely to be readily available (either from literature or from source inventory records kept by water utilities) include:

◗ Freshwater abstraction from the environment (e.g. reduction, due to water recycling);

◗ Energy consumption, predominantly as electrical power (including source and product water transport as well as treatment);

◗ Chemicals consumption by volume and/or mass (for treatment, including transport);

◗ Nutrient discharge to the environment by mass (or reduction thereof) as result of either pre-treatment of the source water for recycling, or as a result of the advanced water treatment process itself;

◗ Waste treatment and disposal volume and/or mass (including transport); and

◗ Global warming potential i.e. Greenhouse gas emissions estimates or 'footprint'.

The calculation of greenhouse gas (GHG) emissions, either from LCA models or simply from electrical energy consumption (predominantly so-called 'Scope 2' emissions), appears to be a commonly used indicator of environmental sustainability. The approach using LCA to estimate greenhouse gas emissions is preferable in that it can be more inclusive by including direct or fugitive emissions (i.e. 'Scope 1') and so-called 'embedded' emissions associated with materials (e.g. chemicals) consumed, waste produced and transport etc. (i.e. 'Scope 3') that simpler GHG accounting methods might ignore.

Even for only one relatively 'simple' estimate of LCA impact potential (e.g. Global Warming), system boundary definition (including GHG accounting method and responsibility) becomes important for meaningful comparison of alternatives and communication to different stakeholders. For example, a water utility planner might be concerned mainly with reporting obligations for Scopes 1 and 2 (and hence carbon tax implications, for example) under the greenhouse reporting protocols such as National Greenhouse and Energy Reporting System (NGERS) in Australia. Alternatively, for a social media dialogue, the complete greenhouse 'footprint' (including all Scopes) of an activity like water recycling might be more relevant. Similarly, issues around uncertainty of GHG estimates will need to be communicated. This aspect was missing from most of the literature reviewed in this study. For example, electrical power from the grid has different applicable GHG emissions factors, depending on location (or State under NGERS) and the degree of power generation from hydro-electric schemes or other

renewable sources contributing to the grid. As another example, if the system boundary for water recycling includes a dam (which has significant potential to emit fugitive methane, and hence GHG), then the associated GHG emissions potentials are uncertain (variable and/or based on limited data) and not currently included under NGERS.

For other LCA impact categories (e.g. ecotoxicity, and human toxicity) there are similar issues of uncertainty that are generally not well addressed in the literature relating to urban water systems, mainly for reasons of complexity. Some of this complexity stems from the limitations or uncertainty associated with the toxicity models themselves that underpin LCA methodology. These aspects may be largely beyond the scope of what can be reasonably here for the NDEEP, except perhaps for comments to a specialised group of stakeholders with a technical interest.

In summary, it is recommended that new LCA models (of a carefully defined case study) should only be built as part this project for the NDEEP if it can be shown that simpler indicators of environmental sustainability are inadequate for demonstration or educational purposes. Such indicators would typically be based on resource, materials and energy inventory items, for example relating to: the use of freshwater resources; energy consumption; use of materials; and discharge of nutrients/ metals/ other potential toxicants. The advantage of rigorous application of LCA methodology is that it attempts to quantify impact potentials as holistically as possible over a wide range of categories spanning ecosystems health, human health and resource depletion. The disadvantages with that such models are: inherently highly case-specific and tied to boundary definition; highly labour-intensive to build and interpret; subject to a significant degree of uncertainty stemming from characterisation and fate factors applied in the impact models used for calculating impact potentials (e.g. ecotoxicity or human health); and relatively complex to communicate to non-technical audiences.

1. Introduction

The Australian Water Recycling Centre of Excellence (AWRCOE) commenced (in December 2011) a project aimed ultimately at developing a National Demonstration Education and Engagement Program (NDEEP). This project has the objective of using the NDEEP to test the hypothetical tenet that: "*Reclaimed water is viewed as an acceptable alternative for augmenting drinking water supplies*".

The project has three streams, each a different but related focus, namely:

◗ Stream 1: Demonstration of water production performance and operational reliability

◗ Stream 2: Social and Governance Research

◗ Stream 3: Developing a National Demonstration, Education and Engagement Programme (NDEEP)

As part of **Stream 1 (Sub-stream 1.4),** GHD has been tasked with undertaking research in the questions relating to **sustainability** of water recycling systems, specifically focussed on providing alternative sources ultimately for the supply of drinking water. It was assumed for this study that water recycling will be used to augment drinking water supplies through indirect means. That is, the recycled water will typically be piped back to the consumer via an impoundment or other source of raw water (e.g. dam, lake or underground aquifer) from which raw water is sourced and drinking water produced.

The overall aim of the research in Sub-stream 1.4 is to test the value of Life Cycle Assessment (LCA) or other similar tools in communicating sustainability trade-offs associated with recycled (or reclaimed) water through the NDEEP. Sustainability itself is complicated since almost all man-made activities impact on the earth's finite resources in a range of different but inter-related ways that have complex mechanisms. Fundamentally, attention to sustainability recognises the fact that the planet has finite resources and that human society will need to live within those constraints. This will require a number of trade-offs to be made, for example, around the water-energy nexus.

The research objective is to assess the value of tools like LCA in communicating, as simply as possible, how the impact potentials associated with urban water supply are spread across a range of categories. Those categories can be broadly grouped to encompass ecosystem health, resource depletion and human health. The primary aim is to demonstrate the use of outputs from tools like LCA as sustainability indicators to guide, in a balanced manner as far as possible, the views taken by stakeholders of recycled water. Those stakeholders include the public, technocrats, planners and politicians. In this manner, the intent is that the trade-offs confronting society around sustainability can be dialogued more effectively.

The purpose of this report is to review literature relevant to the research tasks of Sub-stream 1.4. It addresses the following research question, as defined in the Research Plan for this project:

Question 1: Can we distil sufficient useful information from previous LCA studies (published or otherwise, both in Australian or overseas) to demonstrate the use of LCA outputs as sustainability indicators in a NDEEP for reclaimed water? What are the knowledge gaps or inconsistencies?

This report represents Milestone 1 of the Research plan for this Sub-stream 1.4 for the project.

2. Terminology

As part of the overall Australian Water Recycling Centre of Excellence National Demonstration Education & Engagement Program (NDEEP), the question of terminology surrounding recycled water will be addressed since it can play a critical role in community perceptions. In order to compile this Literature Review, it was necessary to review documentation using a range of terminologies for the product of advanced treatment of water that has passed through wastewater systems in the water cycle. Some of these include:

◗ Water reuse or Reused water

◗ Reclaimed Water

◗ Recycled Water

◗ Purified Recycled Water (PRW)

◗ Potable Water or Potable Reuse

◗ Indirect Potable Reuse

◗ etc.

The preferred term recommended for use in the NDEEP had not been finalised at the time of writing this Review. It was noted that 'Potable Reuse' has some advantages, but might also have the disadvantage that the term 'potable' has the potential to be misunderstood by non-technical persons (e.g. suggesting the need to 'pot' or 'cook'/'boil' the water to be safe).

For the purposes of this Literature Review, the term for the product water used in the original reference was also used here. Therefore several different terms (including those listed above) will appear in this document. The intent was to remain true to the original reference. How if this information is used (if at all) in the development of the NDEEP, carefully consideration of terminology will need to be given by the teams of the respective Streams, during research, development and in the final deliverables.

This review was focussed on literature referring to the product of water treated from a wastewater stream, to a quality standard that is suitable for augmenting drinking water sources, whether by direct means into the supply system intended for human consumption, or indirect means via storages upstream of further treatment prior to distribution through such a system.

3.   Scientific/ technical journal papers

Based on the scope and outcomes of the literature review, the following section provides a summary of relevant published papers from scientific/technical journals. A full list of reviewed papers can be found in Section 7.

3.1   Lundie et al. (2004, 2005) - Sydney

Lundie *et al.* (2004, 2005) applied Life Cycle Assessment (LCA) to the metropolitan water systems planning of greater Sydney. Their study formed part of the review Sydney Water's long-term strategic vision for the region at the time (named *Water Plan 21*). The review required some kind of technical environmental planning tool to compliment the economic and financial tools used in developing the strategic plans. The aim of the study by Lundie *et al.* was to "compare the relative sustainability of operations under different planning scenarios, enabling consideration of the environmental issues in parallel with financial, social and practices considerations in strategic planning". Although their study did not include indirect potable reuse (IPR) as one of the scenarios, it is nevertheless useful examine the work of Lundie *et al.* (2004, 2005) since it was one of the first of its kind to apply LCA for strategic planning in Australia.

3.1.1   Key points from Lundie et al. (2004, 2005)

Several important points of relevance to the NDEEP project here can be observed from the work of Lundie *et al.* (2004, 2005).

Firstly, sustainability issues around the urban water-energy system nexus can most readily be articulated on a *relative basis*. That is, one water supply option (or scenario) can be effectively compared against another when considering sustainability. One option (or scenario) may be assessed, using LCA as a tool, to have a larger or smaller impact potential relative to another scenario. Lundie *et al.* (2004, 2005) compared future scenarios against a base case system model which was constructed to represent Sydney Water's current (at the time) operating assess as augmented to and upgraded to 2021.

To address the question of sustainability in absolute terms (i.e. estimated actual damage[1]) for any man-made activity, such as building or operating towns and cities (including their urban water systems) is a much more complex and subjective task.

Secondly, when comparing urban system planning options, it is important to take a *holistic approach* as far as possible. LCA methodology forces the discipline of defining a system boundary for a given study. For example, to compare seawater desalination as a future (now built) scenario for Sydney, Lundie *et al.* (2004, 2005) needed to include other water supply or energy efficiency options within their system boundary. They defined their system boundary to include the whole urban water system (including existing potable supply from dams, wastewater collection, treatment and disposal plus associated biosolids disposal, along with materials and energy supplies). This enabled desalination to be compared against other water supply or energy efficiency scenarios, including:

---

[1] Damage can be expressed, for example, in terms of loss of human life, Disability Affected Life Years (DALY), loss of ecosystems or depletion of resources.

◗ 'Demand management' (e.g. water restrictions, water-saving devices) that carry with it materials and energy supply savings from avoided consumption.

◗ Energy efficiency within Sydney Water's operations as whole (including transport, pumping systems, lighting etc.)

◗ Energy recovery from biosolids

◗ Local wastewater treatment and water recycling through irrigation for new urban developments.

Clearly, the question of acceptability (or otherwise) of reclaimed water as alternative for augmenting drinking water supplies is likely to raise associated questions in the minds of respondents, such as:

◗ What are the other alternatives for water supply?

◗ What are the relative environmental or health impacts (or impact potentials) of those alternative?

◗ Can the impacts of water supply be minimised in other ways (e.g. by water and energy savings in the total urban water cycle)?

To address these broader associated questions, the work of Lundie *et al.* (2004, 2005) shows that an LCA study needs to include the entire urban water system within its boundary definition. It also illustrates that for the NDEEP, the question of 'acceptability' will need to be framed within a defined urban water system context and scenario, and addressing underlying assumptions, such as:

◗ The urban water system is largely pre-existing (e.g. having evolved from urban infrastructure in Australia cities over the past 50+ years), as opposed to new urban developments with more modern, potentially water-sensitive designs.

◗ Hence, centralised treatment accounts for the majority of current urban wastewater systems and feasible options for use of recycled water by irrigation or other means, either: are limited on a volume basis; or have already been accounted for; or have already been implemented as far as practically possible.

◗ Centralised treatment is a requisite for financially feasible large-scale water recycling to augment drinking water supplies.

◗ Water saving (e.g. 'demand management') measures are already in place, to the limit of practically or long-term public or industry acceptability and economic feasibility. Even with these measures in place, the total system water demand predicates that water recycling to augment drinking water supplies be examined as one option.

◗ Energy saving measures are already in place, to the limit of practicality or long-term public or industry acceptability and economic feasibility. The energy inputs for water recycling cannot be "offset" by saving energy elsewhere in the urban water system, or its associated activities.

3.1.2   Summary from Lundie *et al.* (2004, 2005)

In discussion of the results of their study, Lundie *et al.* (2004) made the following points, which are pertinent to the AWCOE/ NDEEP project:

1. "LCA provides a defensible methodological platform on which to quantify environmental burdens associated with the base case and on which alternative future systems can be compared on a quantitative basis. LCA allows environmental benchmarking of 'business as usual' against promising alternatives for sustainable water services".

2. "Using LCA early on in the planning process helps ensure that environmental issues are considered…. Performing LCA has enabled Sydney Water to capture environmental effects associated with the consumption of materials, which does not routinely occur in other strategic planning processes". (By implication, energy consumption is more readily identifiable, and typically is accounted for in strategic planning).

3. "Building an LCA model on the scale and complexity of an entire large urban water system, such as Sydney Water, is a resource-intensive process".

4. "The process of constructing the LCI involves information exchanges between planning and operational staff which can enhance communication in a large organization. It has also provided information which has enriched communication with external stakeholders".

3.2   Kenway *et al.* (2008) – CSIRO Study

Kenway *et al.* (2008) in a CSIRO National Research flagships study prepared for Water Services Association of Australia (WSAA) focused on the energy used by ten water utilities operating supply and waste water systems in seven cities in Australia and New Zealand, namely: Sydney, Melbourne, Brisbane, Gold Coast, Adelaide, Perth and Auckland. The data collection period was sourced from the water utilities for the 2006-7 financial year. The study was interesting since it provided one of the first attempts to benchmark water-related energy use across Australian cities against each other. The study also benchmarked water and wastewater (supply and treatment) against other societal energy uses in Australian cities, such as residential hot water and society as a whole (i.e. across all sectors of the economy). It provides a context in which to consider recycled (reclaimed) water as an alternative source of drinking water.

3.2.1   Energy intensity and interpretation

The averages of data presented by Kenway *et al.* (2008), when expressed as energy intensity per unit flow of total water supply (kWh/ML) are be summarised in Table 3-1 and Figure 3-1.

Since Kenway *et al.* (2008) presented data for urban population served by the water utilities covered[2], the results can also be expressed as urban water system energy intensities, expressed as average power per capita (W/cap). In these units, the data might be more easy to interpret for the average person who is accustomed to power expressed in Watts (W), such as commonly for light bulbs (e.g. for an average domestic room the power requirement would be 60-100 W for old incandescent bulbs, or 11-21 W for newer energy-efficient bulbs). The results are shown in Figure 3-2.

From the data of Kenway *et al.* (2008), the average total urban water system (as at 2006-7) amounted to approximately 20 W/capita (range 11 to 42 W/capita), which is about equivalent to operating one or two energy-efficient light bulbs per person for 24 hours per day, 365 days per annum. This energy requirement includes pumping and treatment for both water supply as well as wastewater collection and treatment. For the water supply and treatment components only, the average energy requirement amounted to 11 W/ capita (range 1 to 32 W/capita). The large range in the latter can be largely explained from regional differences in water supply source, pumping head (including distance) versus gravity flow

---

[2] For some cities (e.g. Brisbane) the population estimates are only for one of the water utilities (Brisbane Water) in existence at the time and did not include the metropolitan region of greater Brisbane and south-east Queensland region (e.g. Moreton Bay, Redland Bay, Logan). A subsequent study (refer to Section 3.3) collected data from SE Queensland more broadly.

and extent of treatment required (e.g. high pumping energy for Adelaide and both low pumping and treatment energy for Melbourne – refer to Table 3-1).

Table 3-1   Energy intensity of existing urban water systems in Australia/ new Zealand, 2006-7 (from data presented by Kenway *et al.*, 2008, Table 1)

City:	Sydney	Melbourne	Perth	Brisbane	Gold Coast	Adelaide	Auckland	Average
Total energy (kWh/ML *or MWh/GL*)								
Water Supply								
• Pumping	925	85	500	493*	168	1820	91	583
• Treatment	102	9	483	182	39	97	116	147
Wastewater								
• Pumping	66	431	217	128	296	100	114	193
• Treatment	382	694	497	446	706	578	731	576
Other energy demand	137	124	380	158	233	385	62	211
SUM (kWh/ ML water supply)	**1,612**	**990**	**1,537**	**1,233**	**1,101**	**2,512**	**879**	**1,409**

*Brisbane data adjusted by approximation (as per footnote in Kenway *et al.* 2008) to account for pumping energy recorded at the Mt Crosby water treatment plant in the original data
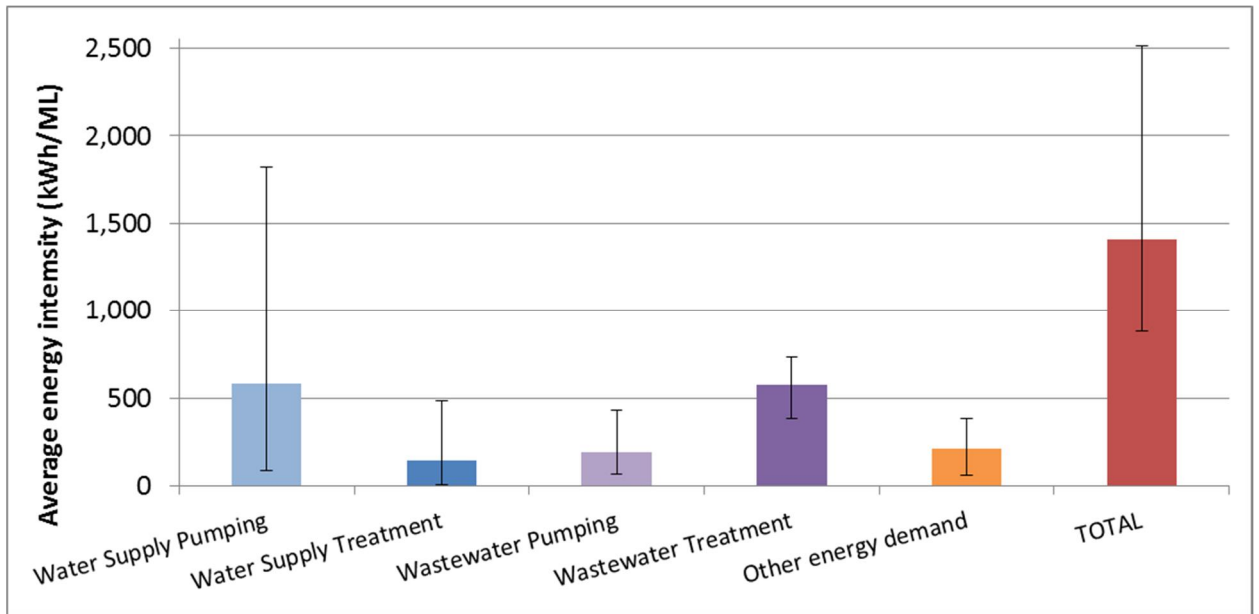
Figure 3-1 Energy intensity per megalitre water supply for existing urban water systems (2006-7) in Australia and New Zealand (from data presented by Kenway *et al.,* 2008, Table 1). Error bars denote range of annual averages across cities represented by water utilities surveyed.
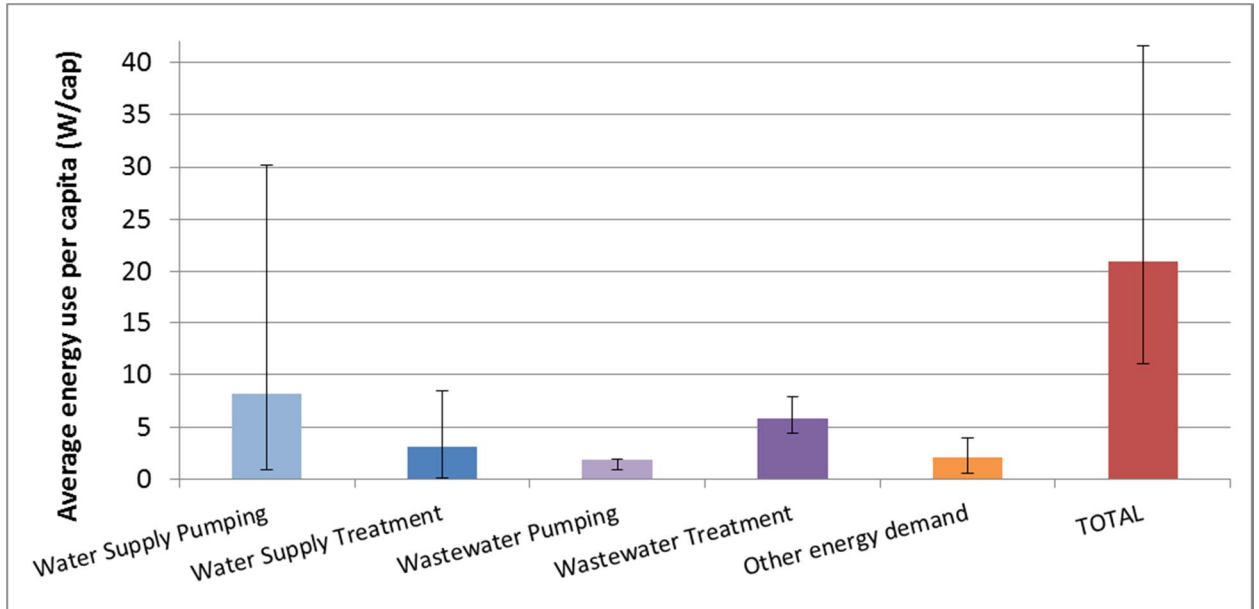
Figure 3-2 Energy intensity per capita for existing urban water systems (2006-7) in Australia and New Zealand (from data presented by Kenway *et al.,* 2008, Table 1). Error bars denote range of annual averages across cities represented by water utilities surveyed.

Kenway *et al.* (2008) compared the energy use of water utilities (i.e. total urban water systems existing in 2006-7) to that for residential hot water systems in Australia and the urban system as a whole. The

estimate for the urban system as a whole includes the total Australian economy, pro-rated per city[3], including agriculture, manufacturing/ construction, transport and mining. Again, using the data from Kenway *et al.* (2008) and converting from units of MJ/capita to W/capita for ease of understanding, the results are summarised in Figure 3-3. The log scale of the y-axis should be noted.

	Sydney	Melbourne	Brisbane	Gold Coast	Perth	Adelaide	Weighted average
Water Utility	20	12	14	14	23	37	19
Res. hot water	103	131	94	129	124	174	121
Urban system	7071	9006	9957	9957	12747	6786	8784
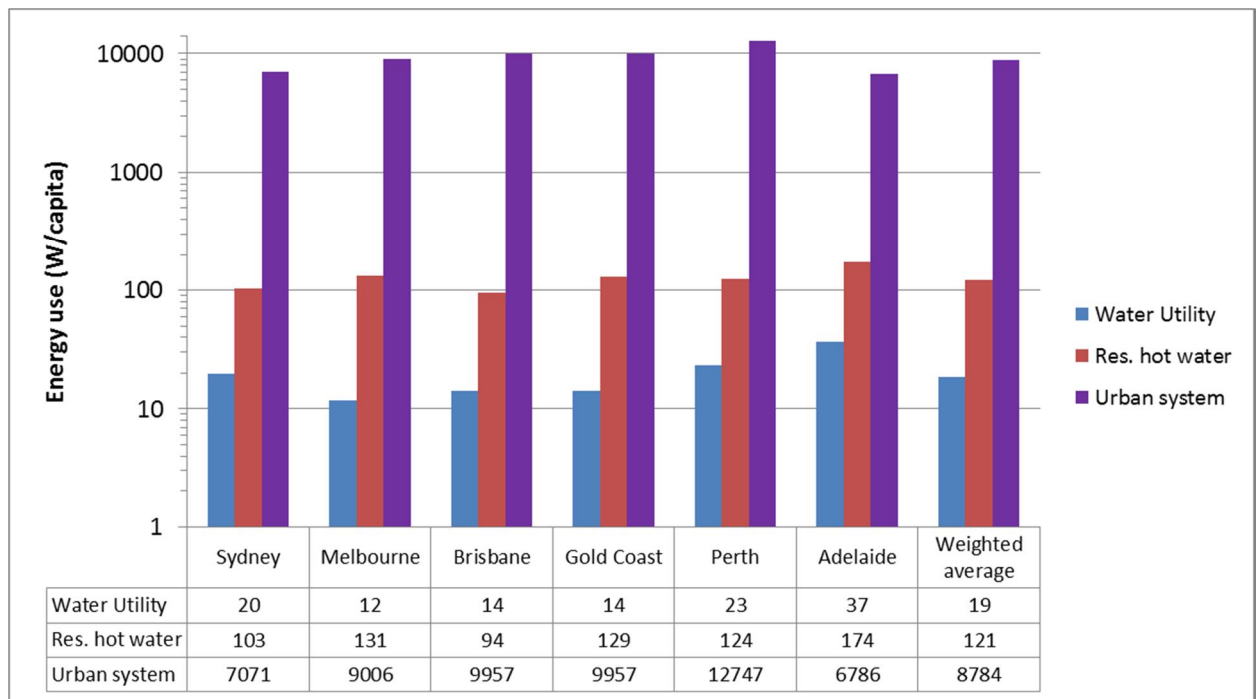
Figure 3-3   Comparison of energy consumption per capita for water utilities (water supply and wastewater collection/ treatment) with that for residential water heating and the societal urban water system as a whole (from data presented by Kenway *et al.*, 2008, Table 8)

The weighted average for the urban system in 2006-7 compares well with the results published by the Australian Bureau of Statistics[4] (ABS) and Australian Bureau of Agricultural and Resource Economics[5] (ABARES):

◗ Total population (Australia, 2006 census data): 20,848,760 capita

◗ Total energy consumed (sum all sectors of economy): 5,770 PJ per annum

◗ Total energy consumed per capita = 2.77E-04 PJ/capita = 8776 W/capita

From Figure 3-3 it is obvious that energy use by water utilities (in 2006-7) in Australia typically represented about 0.2% of societal (or urban system) total energy use and about one-sixth (around 15%) of that used for residential hot water. An understanding of such relativities might help to provide a 'sanity'

---

[3] Kenway *et al.* (2008) noted that their approach assumes that the individuals in each city have influence over energy consumption in the rest of the state and hence the country as whole. The results are therefore indicative for comparative purposes only.

[4] http://www.abs.gov.au/AUSSTATS

[5] http://www.daff.gov.au/abares/publications_remote_content/publication_series/

check when communicating energy requirements and sustainability for water supply alternatives. Kenway *et al.* (2008) drew attention to the potential savings from water consumption or hot water heating (including possible energy efficiency improvements) to provide offsets for increased energy requirements from alternative water supply from such as desalination.

3.2.2   Greenhouse gas intensity

The energy intensity data for urban water supply systems from Kenway *et al.* (2008) (Section 3.2.1) can be converted to greenhouse gas (GHG) emissions on the assumption that (virtually) all the energy used in water systems is sourced as electricity purchased from the grid (i.e. reported largely as 'Scope 2' indirect emissions in GHG accounting protocols). Recent emission factors for Scope 2 (electricity purchased from the grid) in Australia and New Zealand are given in Table 3-2.

Table 3-2   Greenhouse gas emission factors for Indirect (Scope 2) emissions for electricity purchased from the grid in Australian states and Auckland (New Zealand)

City or State	Scope 2 emission factor adopted (kg CO2-e/kWh):
Sydney (NSW)	0.89
Melbourne (VIC)	1.21
Perth (WA)	0.88
Brisbane (QLD)	0.88
Gold Coast (QLD)	0.8
Adelaide (SA)	0.68
Auckland (New Zealand)	0.137

Sources: DCCEE (2011) and MFE (2011)

It is worth noting from Table 3-2 that the emission factors for New Zealand are significantly lower, mainly due to a major part of the electricity grid supply in that country being from hydro power stations. So as not to skew the perspective for Australia, Auckland was excluded from the GHG calculations here (see below).

Multiplying the energy intensity data per megalitre (ML) of water supply (for each of the Australian city water utility data in Kenway *et al.,* 2008) as summarised in Figure 3-1, by the relevant State-based emission factors in Table 3-2, the GHG intensity graph in Figure 3-4 can be derived. Note that the uncertainty range (denoted by error bars) changes since the variation in state-based emission factors is compounded with the variation in energy intensities. The range for water supply (pumping plus treatment components) is 0.11 to 1.30 tonnes CO_2-e per ML water supply (average 0.66), excluding Auckland.
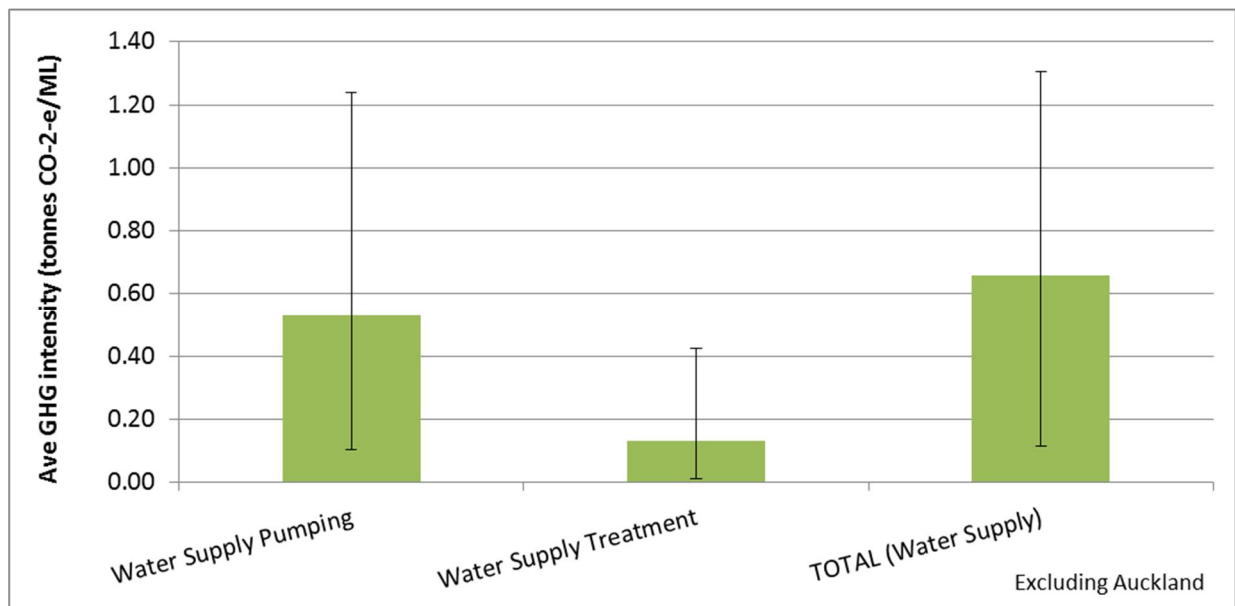
Figure 3-4 Greenhouse gas (Scope 2 emissions only) intensity per megalitre of water supply for water utilities supplying major Australian cities, derived from the data of Kenway *et al.* (2008 – Table 1).

3.3   Cook et al. (2012) – CSIRO Study Update

This study was a follow-up to that of Kenway *et al.* (2008) and providing updated energy data from major water utilities in Australia for the 2009-10 financial year. The cities covered in the study by Cook *et al.* (2012), were Sydney, Melbourne, South East Queensland (centred on Brisbane), Perth, Canberra, Adelaide and Newcastle. Unlike Kenway *et al.* (2008), Auckland was not included in the study by Cook *et al.* (2012). A summary of energy intensity data calculated from the Cook *et al.* (2012) data is given in Table 3-3.

Based on the data provided by Cook *et al.* (2012), the parallel graphs to those in Figure 3-1 and Figure 3-2 are shown in Figure 3-9 and Figure 3-7. By comparing these two figures, it interesting that on average the total energy for urban water systems went down by a small margin (-7%) for the cities considered, but this obscures the fact there was a large variation across cities. According to Cook *et al.* (2012), "*overall the data highlights that the biggest changes in energy use in water services have occurred in water supply and not in wastewater. This reflects the focus on water supply security over the last few years during drought which has seen more reliable sources added to the mix including climate-independent sources such as desalination and water recycling. These do require higher levels of treatment but on the positive side they are sources for our cities that can be developed close to where the water is needed, thus reducing pumping costs*".

A more detailed breakdown of water supply (pumping and treatment) according to cities is given in Figure 3-8, contrasting data 2006-7 with 2009-10. In some cases the trade-off between pumping and treatment energy is evident (e.g. Sydney). In other cases both pumping and treatment energy have increased, coming off a low base (e.g. Melbourne) or relatively high base (e.g. Perth); and in yet other cases, treatment energy has changed little, but pumping costs have come down (e.g. SEQ/ Brisbane).

These differences are obviously highly regional, being affected by topography, catchment characteristics, yield, pumped supply distances and head, proportion of water supplied by desalination and seasonal rainfall variations.

Importantly, then, in the context of the NDEEP and this project it is difficult to generalise about the relative merits of water recycling as an alternative supply. The 'base case' against water recycling is compared (e.g. on energy intensity) will vary from location to location, supply system design and operation, and even from year to year for a given location, depending on rainfall and yield from existing catchments.

Greenhouse gas intensity trends (comparing averages for the same cities in Australia, from the data of Cook *et al.,* 2012 and Kenway *et al.,* 2007) are shown in Figure 3-5. The trend appears to be slightly downwards and with less variability in 2009-10, compared with 2006-7. However, for reasons discussed (see above), this is largely dependent on a number of regional and seasonal factors that influence energy intensity of the urban water supply mixes for the major Australian cities surveyed.
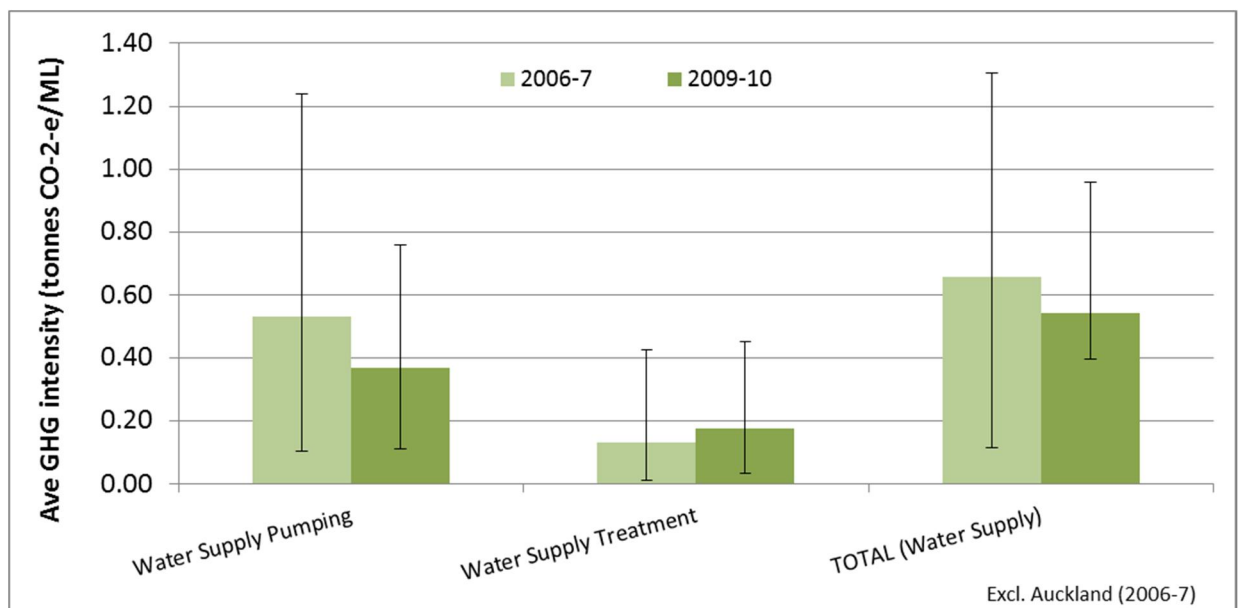
Figure 3-5   Greenhouse gas (Scope 2 emissions only) intensity per megalitre of water supply for water utilities supplying major Australian cities, derived from the data of Kenway *et al.* (2008) and Cook *et al.* (2012).

Table 3-3   Energy intensity of existing urban water systems in Australia, 2009-10 (from data presented by Cook *et al.*, 2012, Table 1)

City:	Sydney	Melbourne	SEQ incl. Brisbane	Perth	Canberra	Adelaide	Newcastle	Average
Total energy (kWh/ML *or MWh/GL*)								
Water Supply								
• Pumping	284	299	126	865	328	672	410	426
• Treatment	278	75	511	226	167	106	36	200
Wastewater								
• Pumping	66	168	198	212	32	96	179	136
• Treatment	425	686	590	616	813	813	452	628
Other energy demand	121	152	173	261	456	425	477	295
SUM (kWh/ ML water supply)	**1,103**	**1,181**	**1,093**	**1,604**	**1,276**	**1,517**	**1,354**	**1,304**
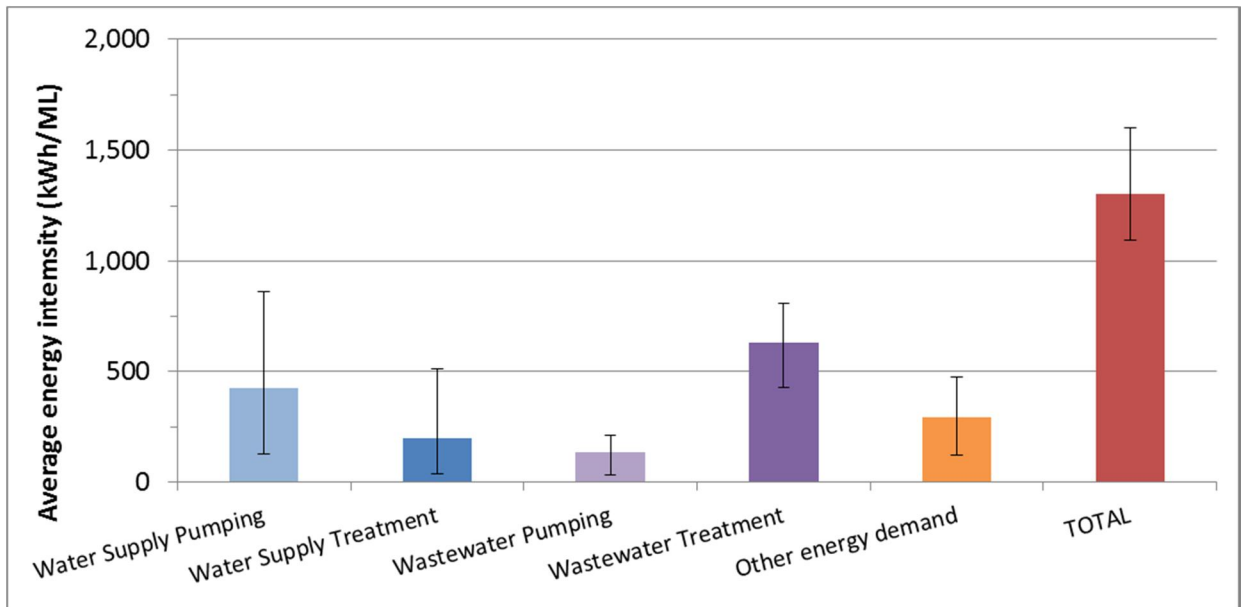
Figure 3-6 **Energy intensity per megalitre water supply for existing urban water systems (2009-10) in Australia (from data presented by Cook *et al.,* 2012, Table 1). Error bars denote range of annual averages across cities represented by water utilities surveyed.**
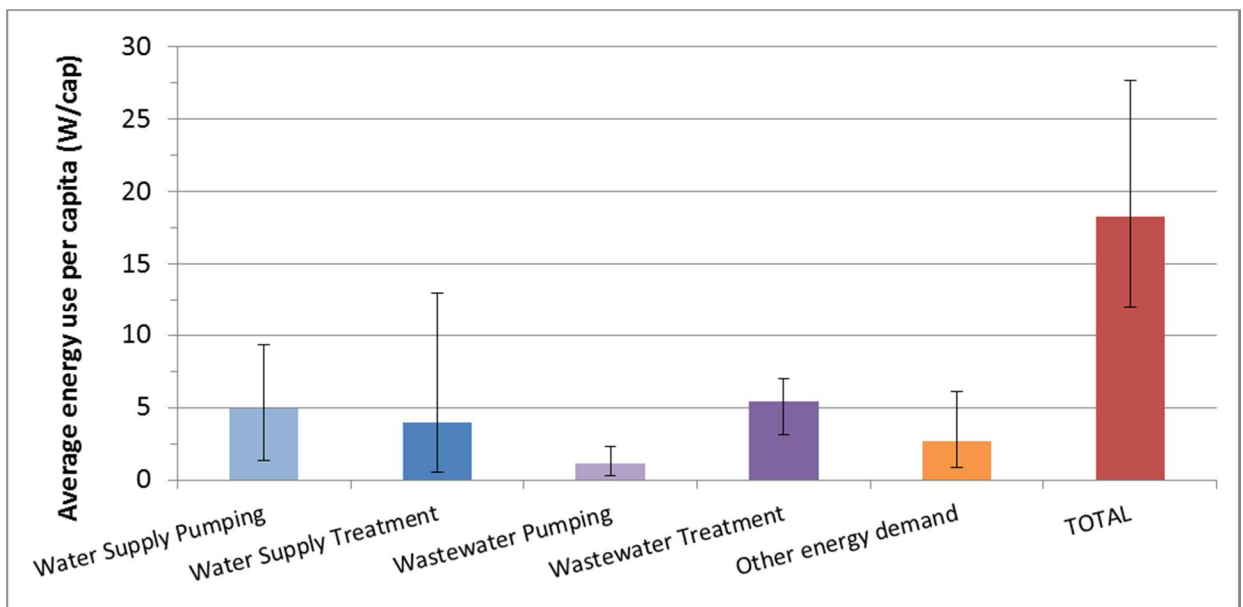
Figure 3-7 **Energy intensity per capita for existing urban water systems (2006-7) in Australia (from data presented by Cook *et al.,* 2012, Table 1). Error bars denote range of annual averages across cities represented by water utilities surveyed.**

Water supply - Pumping

	Sydney	Melbourne	SEQ/ Brisbane	Perth	Canberra	Adelaide
2006-7	925	85	500	493	168	1820
2009-10	284	299	126	865	328	672

Water supply - Treatment

	Sydney	Melbourne	SEQ/ Brisbane	Perth	Canberra	Adelaide
2006-7	102	9	483	182	39	97
2009-10	278	75	511	226	167	106
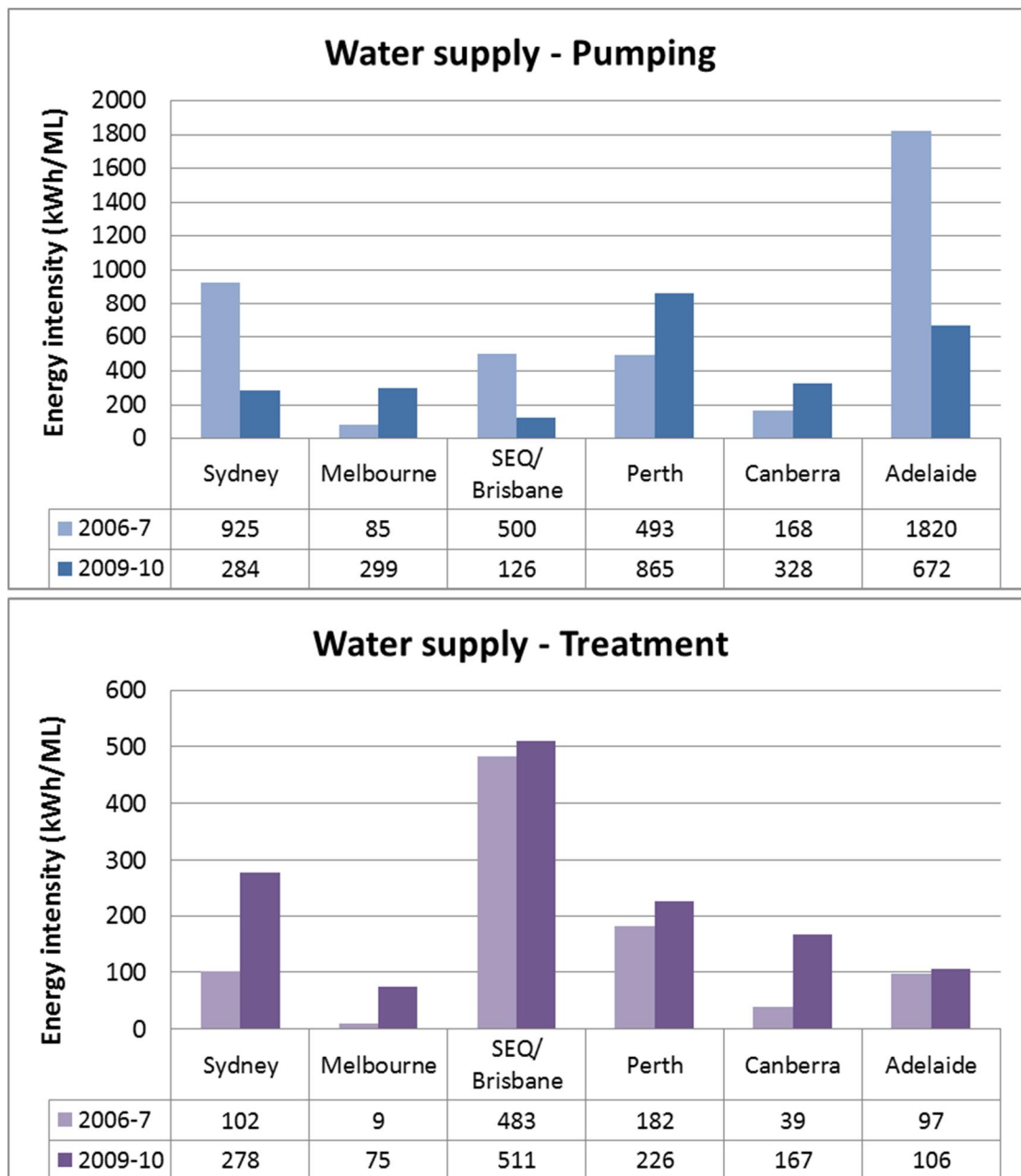
Figure 3-8   Water supply and treatment energy intensity for existing urban water systems in Australia – comparison of data for 2009-10 (Cook *et al.,* 2012) and 2006-7 data (Kenway et al., 2008)

3.4 Hall *et al.* (2009) – UWSRA Study. SE Queensland

As part of the Urban Water Security Research Alliance in Queensland, Hall *et al.* (2009a,b) studied energy and greenhouse gas emissions for the SEQ Water Strategy. This strategy (prepared by the Queensland Water Commission) incorporated a number of future scenarios with population and water demand projections, including how that demand would be met by a combination of existing and new infrastructure. A significant part of that new infrastructure has subsequently been built in the form of the South East Queensland (SEQ) Water Grid, the Western Corridor Purified Recycled Water Scheme and the Tugun Desalination Plant treating seawater.

3.4.1 Energy intensity

Hall *et al.* (2009a) collected data on the operations of centralised water supply systems for SEQ from a number of sources, including reports to the Queensland Water Commission, SEQ utility surveys (Kenway *et al.,* 2008) and reports for SEQ grid energy performance (Jacob and Whiteoak, 2008). Based on available data, Hall *et al.* (2009a) calculated the energy intensity for a range of water supply options identified in the SEQ Water Strategy. These included a number of Advanced Water Treatment Plants (AWTPs) to produce Purified Recycled Water (PRW). PRW can potentially be recycled to the urban tap as drinking water via a dam or other impoundment (i.e. for an indirect potable reuse scheme). The Western Corridor PRW scheme is one such example (now built), which was included in the data of Hall *et al.* (2009a), along with other potential similar schemes in SEQ.

It is useful to compare the data for nine[6] PRW schemes considered by Hall et al. (2009a) with that for major existing water utility water supply systems in Australia and New Zealand at the time (Kenway *et al.*, 2008 – refer to Section 3.2 above). The existing systems at that time were based largely on traditional water sources (dams and/or rivers, supplemented with groundwater in some cases), except for Perth where a seawater desalination plant was commissioned in 2006. The comparison is given in Figure 3-9.

Figure 3-9 illustrates that the average energy intensity of PRW schemes might be expected to be typically about three times higher than that for exiting systems. In an extreme case, the difference might be as much as twenty-four times higher. At best, the two alternatives (existing vs. PRW) might be comparable in terms of energy intensity. The large range stems particularly from the regional differences in energy intensity of existing systems from the 2006-7 source data (e.g. Melbourne's existing predominantly gravity-fed water supply from protected catchments with only limited treatment; vs. Adelaide or Perth's supply with a large pumping and/or treatment component, including desalination in the case of Perth at the time). Also, PRW schemes differ particularly in terms of the energy intensity for treatment (e.g. membrane vs. non-membrane systems) as well as pumping (e.g. depending on the head requirements for pumping to the dam at the upstream end of the catchment).

---

[6] The Western Corridor PRW Scheme now exists but is currently not fully operational, while the remainder were potential future or hypothetical in the QWC Strategy examined by Hall et al, (2009)
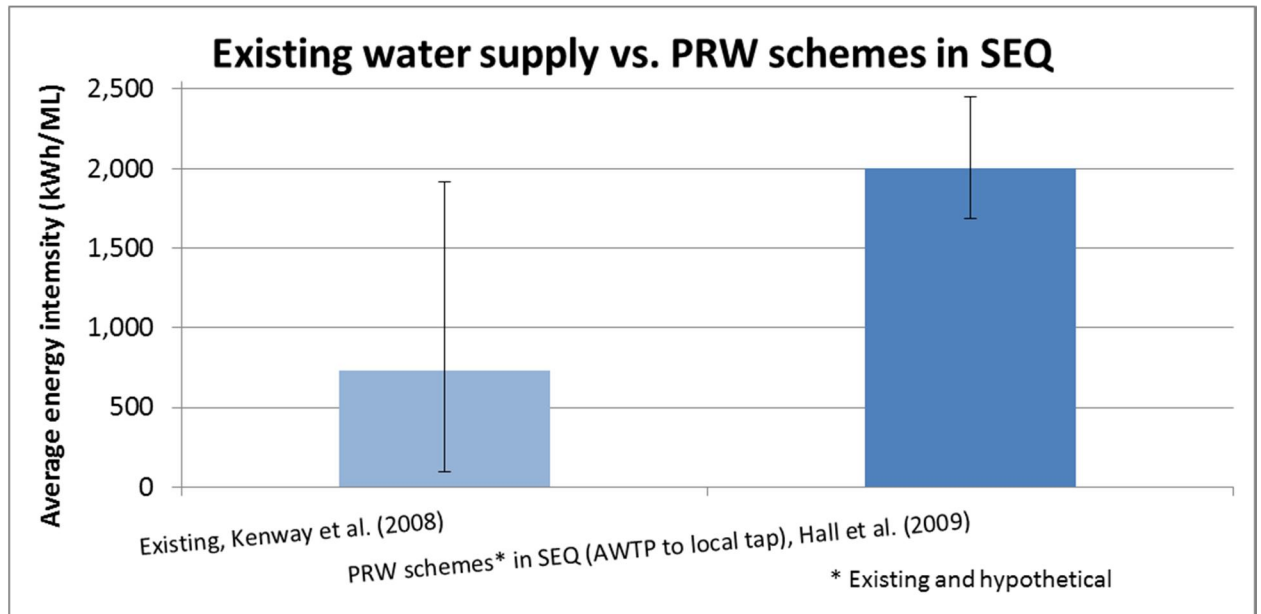
Figure 3-9   Comparison of energy intensity for existing urban water supply systems by water utilities for major cities in Australia/ New Zealand (Kenway *et al.*, 2008) with that for alternative PRW schemes in SE QLD (Hall *et al.,* 2009a). Error bars denote range of annual averages across cities represented by water utilities or schemes considered.

3.4.2   Greenhouse gas intensity

Hall *et al.* (2009b) estimated the greenhouse gas intensity for the urban water supply alternatives, based on the infrastructure mix defined by the SEQ Water Strategy for the next 50 years. Hall et al. (2009) assumed a Scope 2 emissions factor of 1.04 kg CO2-e/kWh for electricity purchased from the grid in Queensland. Since the updated emissions factor for this state is lower (0.88 kg CO2-e/kWh, refer to Table 3-2), the estimates from Hall *et al.* (2009b) can be adjusted downwards by 15%. An extract of data for water supply (mainly from traditional dam sources) and purified recycled water (indirect potable reuse) based on data from Hall et al. (2009b) is given in Table 3-4.

The emission intensity for the centralised SEQ-wide water supply (traditional) in Table 3-4 fits in the mid-range of averages that can be calculated on the same basis from the 2006-7 data of Kenway *et al.* (2008) (Table 3-1) for pumping and treatment, namely:

◗   Brisbane: 0.59 tonnes CO2-e/ ML (675 kWh/ML)

◗   Gold Coast: 0.18 tonnes CO2-e/ ML (208 kWh/ML)

On the basis of the data in Table 3-4, the GHG emissions intensity of PRW supply is about four times higher than that of traditional water supplies in SEQ.

Table 3-4 Summary of Greenhouse gas emission intensity data sources and uncertainty based on data of Hall *et al.* (2009b), after adjustment for Scope 2 emissions factor**

Emission source	Emission intensity** (tonne CO_2-e/ML)	Interval from mode	Data accuracy	Summary of data sources
Indirect (energy) for centralized water supply	0.35 (0.30 pumping; 0.045 treatment)	±15%	good	SEQ utility surveys and reports for SEQ grid energy performance
Indirect (energy) for purified recycled water supply	1.52 (0.73 pumping; 0.83 treatment)	±15%	good	As above but not all available plants required for the supply mix

** Adjusted for Scope 2 emissions factor = 0.88 kgCO2-e/kWh in QLD (DCCEE, 2011)

3.5   Sherman et al. (2001) - Dam methane

Hall *et al.* (2009a) reviewed greenhouse gas emissions form reservoirs (i.e. lakes and dams). They cited extensively from the work of Sherman (2001) and Sherman *et al.* (2001). Some key points from the review by Hall *et al.* (2009a) may be listed as follows:

◗ Most Australian reservoirs more than 6 or 7 m deep are persistently thermally stratified during spring through autumn since absorption of solar radiation in the water column causes the surface waters to warm more than the deep waters. This stratification suppresses vertical transport in the water column to the extent that the interior of most reservoirs are quiescent. A consequence of this stratification is that dissolved oxygen becomes depleted in deeper waters (the hypolimnion) due to respiratory demands and carbon dioxide (CO_2) accumulates. When dissolved oxygen is effectively exhausted (a common occurrence) then methane (CH_4) might accumulate as well.

◗ The important additional contributions to global warming due to the presence of reservoirs arise from two factors: a one-time breakdown of soil and plant carbon as a result of inundation when a storage fills; and on-going emissions of methane rather than CO_2 because conditions in reservoirs often promote anaerobic conversion of organic carbon to CH_4 rather than CO_2 both in the water column and in the sediments. Implicit in this assessment is the assumption that reservoir methane emissions would have occurred instead as CO_2 (i.e. greenhouse neutral) had these taken place in a natural river channel or in the ocean.

◗ Organic carbon in reservoirs arises from a number of sources, including plant material deposited in the littoral zone (i.e. the edges either at inundation or from growth at the verges when water levels drop) or deposited in the sediments from plant growth in the water column (e.g. by photosynthesis and death of algae and macrophytes).

◗ Organic carbon is converted to methane under anaerobic conditions in the sediments due to a combination of fermentation and methanogenic bacterial activity. The methane can either diffuse through the water column or bubble to the surface and be released to the atmosphere as gas. Alternatively, a fraction of the methane formed will dissolve in the water column (due to supersaturation at depth) and might be released to atmosphere when water is discharged from the reservoir. It is also known that a significant part of the dissolved methane can be oxidised to CO_2 in

the aerobic upper layers of the water column by methanotrophic bacteria, in which case it becomes greenhouse neutral.

◗ (Net) Methane emission rates have been measured by sampling and analysis of gas emissions at the surface of reservoirs. The review included studies from Canada, Scandinavia, South America and Australia. In tropical climates, reservoir methane emission was found to be much higher than observed in colder boreal (forested) regions. Emission rates are also indicatively four to 12 twelve times higher in the first year following inundation (i.e. when the reservoir is first filled), compared with the fourth year.

◗ The available data for methane emissions from Australian reservoirs is limited (Table 3-5). The emission rates are highly variable (range spanning two orders of magnitude) but median values in the range 40 to 220 mg/(m^2.d) compare well those of locations in other similar tropical parts of the world (French Guiana and Brazil) – compare Table 3-5 and Table 3-6. Higher values in the range 1000 to 1760 mg/(m^2.d) have been recorded in Australia but the data set is too small to be certain how representative these values are of typical emission rates.

From the perspective of the AWRCOE project here, reservoir (e.g. dam) methane emissions will be relevant from a greenhouse gas point of view if a dam is included in the boundary for sustainability or life cycle assessment. For example, the traditional urban water supply system would typically include a dam and might be the reference point for educational awareness of the sustainability impacts of water recycling by indirect potable reuse (IPR). Moreover, if the IPR system recycles water to a dam, thereby augmenting 'raw water' supplies and avoiding the need for building a new dam (or increasing the area occupied by one or more existing dams), then avoided dam methane emissions should be taken into account when discussing sustainability issues.

Table 3-5 Measured methane emission rate (flux) from Australian reservoirs

Location	CH_4 flux (mg/(m^2.d))		
	Min.	Median	Max.
Wivenhoe (n > 8)	24	40	73
Borumba (n = 1)	-	80	-
Little Nerang (n = 3)	-	1,000	-
Chaffey Dam (n=2)	38	220	1760

Source: Hall *et al.* (2009a), Table A6.2

Table 3-6 Measured methane fluxes from reservoirs in boreal and tropical climates

Location	CH_4 flux (mg/(m^2.d))			
	Mean	Median	Min.	Max.
73 mostly small (<100 ha) northern hemisphere lakes	26	6.8	0.03	162
2 pairs of shallow ice-coverd Finnish lakes. One pair oxygenated; one pair natural	11.5 0.16 (oxy) Note 1	-		-
Lokka Reservoir (1994; 1995)	9.3; 25	16; 12	-0.4; -6.5	48; 244
Porttipahta	2.6	3	-0.5	7.6
Petit Saut	260	-	-	-
Tucurui	67 ± 45	-	-	-

Source: Hall *et al.* (2009a), Table A6.2

Lokka & Porttipahta are hydo-electric reservoirs in Finland

Petit Saut is a tropical reservoir located in French Guiana (South America)

Tucurui is a reservoir located in Brazil

Note 1: Original data given in CO2-e. Assuming Global Warming Potential (CO2-equivalence) of methane: 21 gram CO2-e per gram methane (DCCEE, 2011)

3.6   Poussade *et al.* (2011) – SE Queensland

Poussade *et al.* (2011) used LCA methodology to compare the energy (power) consumption and greenhouse gas emissions of two major alternative water supply plants in SE Queensland: the Tugun Seawater Desalination Plant, situated at the Gold Coast, supplying the SEQ Water Grid; and one of the Advanced Water Treatment Plants (AWTP), situated at Bundamba near Ipswich, for supplying purified recycled water (PRW) to the Western Corridor scheme for potable reuse. The Tugun Desalination Plant has a capacity of 125 ML/d and the Bundamba AWTP has a capacity of 66 ML/d.

3.6.1   Electricity intensity

In 2009-2010, the average daily water production was 21.9 ML/d out of a 66 ML/d capacity at Bundamba AWTP, and 63 ML/d out of a 125 ML/d capacity at the Gold Coast desalination plant. Seawater desalination production and distribution required a total of 3.82 kWh/m^3, while PRW production and distribution came to a total of 1.73 kWh/m^3. When considering the treatment process alone, the electricity requirement was reduced to 3.30 kWh/m^3 and 1.14 kWh/m^3 respectively for desalination and water recycling. Figure 3-10 provides a breakdown of the electricity consumption per process step on the desalination plant (a), and recycling plant (b), relative to the total electricity used for production and distribution of water. Most of the energy (84%) is used for the reverse osmosis (RO) step in seawater desalination, and it also dominates the single largest energy consumption component in the AWTP for water recycling, although the overall energy requirements are lower and more distributed in the latter.
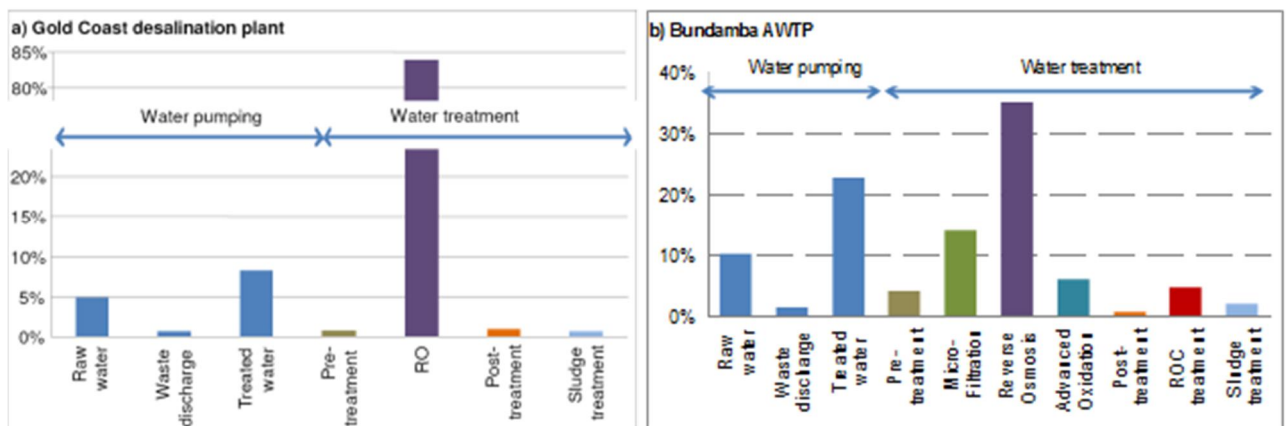
Figure 3-10 Electricity consumption per process step: desalination plant (a) and water recycling plant (b). From Poussade *et al.* (2011)

Both the desalination plant and the AWTP showed limitations in the achievable energy 'turn-down' when operating at less than full capacity. For example, operating the desalination plant at 33% results in a 10% higher electricity consumption per m^3 of treated water produced when compared to operating at between 66% and 100%, which can be explained by the internal by-pass of seawater required for reaching the appropriate velocity in the discharge diffusers. In the case of the AWTP, operating at 15 to 23 % of design capacity gave electrical energy consumption of ~1.3 kWh/m^3 whereas at approximately 45% of design capacity, the electricity consumption was ~1.0 kWh/m^3 (see Figure 3-11).
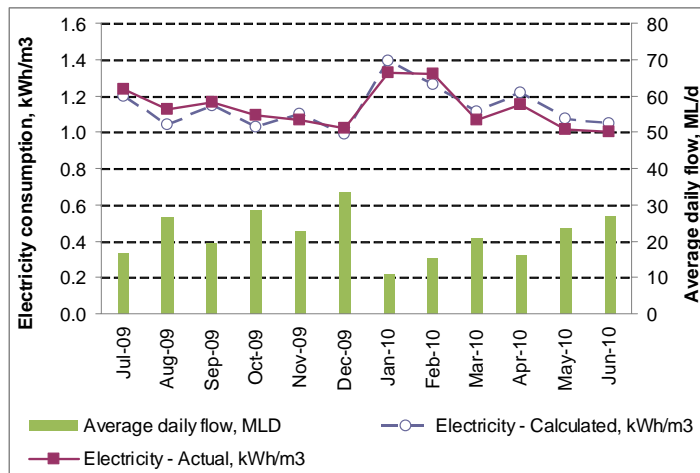
Figure 3-11 Bundamba AWTP – electricity consumption (AWTP only) as a function of production flow. From Poussade *et al.* (2011)

3.6.2 Greenhouse gas (GHG) intensity

GHG emissions from Bundamba AWTP and the Gold Coast desalination plant are shown in Figure 3-12. Production and distribution of desalinated water generates just over twice the amount of GHG generated from water recycling (4.2 and 2.0 kg CO_2-eq/m^3 respectively). In both cases, the production of electricity used by the plants represents a major contributor to GHG emissions, 85% and 95% respectively for water recycling and desalination. This is followed by chemicals production and plant and pipe network construction. The purchase of Renewable Energy (e.g. from solar energy) for the desalination plant in order to offset the emissions of GHG from the purchase of electrical energy is also shown in Figure 3-12.
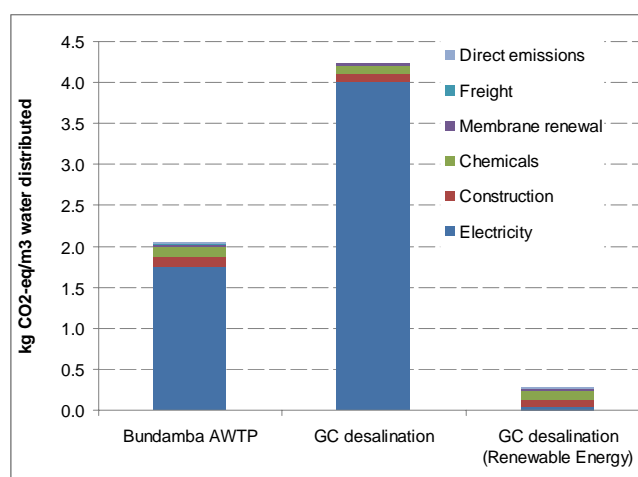
Figure 3-12 GHG emissions (CO_2 equivalents) – Comparison of water recycling and seawater desalination. From Poussade *et al.* (2011)

It is worth noting that the GHG emissions recorded in Figure 3-12 for the AWTP are similar to those estimated by Lane *et al.* (2011) (refer to Section 4.1.5 and Table 4-1 below) for Indirect Potable Reuse. In their case study of the Gold Coast, Lane *et al.* (2011) based their data for a hypothetical AWTP on the a similar operating period (2009-10) of the Bundamba AWTP (data supplied by Poussade and co-workers), adjusted for differences in energy requirements for treated water pumping.

It is also worth noting that the GHG emissions recorded in Figure 3-12 are in the order of 2.8 to 5.8 times greater than that estimated by Lane *et al.* (2011) for a conventional water supply system that is dam sourced - refer to Section 4.1.5 and Table 4-1 below.

3.7 Friederich *et al.* (2009a,b) – Durban (South Africa)

The background to these studies can be summarised as follows (Friedrich *et al.*, 2009a):

"*In the eThekwini Municipality (Durban) potable water is sourced from two impoundments (Inanda Dam and Nagle Dam) and treated in two water treatment plants (Wiggins and Durban Heights Waterworks) after which it is reticulated to the consumers. The outer peri-urban regions of the municipality have on-site sanitation disposal. In the central area and in the suburbs of the municipality, a sewer system collects the used water, which is treated in a number of sewage treatment plants prior to final discharge into the Indian Ocean. A water recycling plant, commissioned in 2001, takes treated sewage to produce industrial grade water, which is used in a paper mill and in an oil refinery. This reduces the demand for potable water in the municipality, freeing this water to be supplied to previously un-serviced households. In this paper, LCA studies were employed to explore the environmental burdens of supplying potable water and sanitation in the eThekwini Municipality and a series of scenarios were modelled in order to find the best environmental options for increasing supply. An important question was related to the recycling operation and its associated environmental burdens.*"

Although these studies did not include PRW (i.e. potable or IPR) schemes, they are still significant in that the authors applied carbon footprint and LCA methodology to complete urban water systems (i.e. similar to the work of Lundie *et al.* for Sydney – refer to 3.1).

3.7.1 Friederich *et al.* (2009a)

The most relevant points from Friederich et al (2009a) for the AWRCOE project here can be summarised as follows:

◗ Water abstraction from the major dam (Inanda Dam) supplying the Durban metropolitan area requires significant pumping, at an energy intensity of 0.240 tonnes CO2-e/ ML water supplied. (Note that this average is of the same order as water supply pumping in Figure 3-9 (see above), based on the data of Kenway *et al.* (2008) and Cook *et al.* (2012) for some major Australian water utilities).

◗ The water recycling plant (WRP) studied (supplied with secondary effluent from the Durban Southern WWTP located adjacent to it) was based on advanced treatment processes without membrane technology. The processes used in the WRP were: chemical coagulation/ settling (lamellae)/ dual media filters/ ozone/ granular activated carbon/ chlorination. The WRP supplied recycled water (approx. 40 ML/d) of non-potable quality to industrial customers also located nearby. The main industry customers were a paper production plant and an oil refinery.

- The absence of membrane technology in the WRP, along with the use of energy-efficient treatment process units (e.g. type of ozonator) meant that its greenhouse gas intensity was relatively low i.e. average of 0.101 tonnes CO2-e/ ML water supplied.

- A large water treatment plant for the city of Durban (Wiggins Water Works; capacity 350 ML/d) uses a similar process flow-sheet but was found to be somewhat less greenhouse gas efficient with an average of 0.185 tonnes CO2-e/ ML water supplied. The authors put this down to less efficient ozonation as the major power consumer of the treatment process units, but highlighted that this required more investigation. They assumed that a more greenhouse (i.e. energy) efficient design for a water treatment plant using a similar process flow-sheet could achieve an energy intensity of 0.130 tonnes CO2-e/ ML water supplied i.e. 30% less than the existing Wiggins plant. Note that this would make a new plant of that type comparable to the averages shown for water supply treatment in Figure 3-5 (see above), based on the data of Kenway *et al.* (2008) and Cook *et al.* (2012) for major Australian water utilities.

- Three options were examined for increased water supply, plus associated wastewater collection and treatment, to a projected 200,000 new households (52 ML/d water supply). The options were:
  - Option 1: Maximise existing assets (water supply from dam with increased pumping and throughput of existing water treatment plant)
  - Option 2: Water recycling (increased water recycling from new or existing wastewater flows generated for use by industrial customers to free up existing potable water supplies, which will serve new household connections)
  - Option 3: Construct new infrastructure (new dam, new waterworks, distribution, sewer collection and treatment systems).

- Of the three options, water recycling had the lowest global warming potential (lowest estimated greenhouse gas emissions) by a margin of about 15% compared to the next-lowest option (Option 1).

- The analysis did not include the potential for greenhouse emissions (notably methane) to be generated in the dam itself (refer to Section 3.5 below). This would have made Option 3 (new dam) even less attractive.

One of the conclusions reached by Friederich *et al.* (2009a) was as follows:

"Water recycling should be encouraged in cases where it is environmentally efficient. The water recycling should take place preferably close to large industrial customers who can accept a lower quality of water as the environmental burden of getting recycled water up to a standard for human use could be high. An important conclusion that is applicable to this analysis is that 'if there is no need to use potable water then don't supply it'. Water recycling may become environmentally inefficient if the distance between the recycling plant and the user is considerable and the pumping requirements are increased and/or if the quality of the incoming water deteriorates to the extent that more energy intensive treatment processes are needed. Therefore, each new recycling initiative should have an LCA study undertaken."

To this might be added that high water quality requirements for the product from water recycling plants (e.g. for indirect potable reuse purposes to augment drinking water supplies) will also predicate additional treatment steps and most likely significantly increase energy intensity. This is illustrated by comparing the relatively low greenhouse gas intensity for treatment in the Durban Southern water recycling plant (Friederich (2009a - see above) with that for treatment to PRW quality for indirect potable reuse (Hall et al, 2009b – see Table 3-4 above), which is about eight times higher.

3.7.2 Friederich *et al.* (2009b)

In this second paper, the authors performed a more complete LCA model to the same systems and scenarios/ options defined by Friederich (2009a). Friederich (2009b) used the CML methodology (from The Netherlands) for their LCA analysis, thereby encompassing a wider range of impact categories than only global warming (greenhouse gas emissions from energy consumption) as in the previous paper. The (midpoint) impact potential categories used were:

- Global Warming
- Ozone Depletion
- Acidification
- Eutrophication
- Photo-oxidant (i.e. smog) Formation
- Aquatic Ecotoxicity
- Terrestrial Ecotoxicity
- Human Toxicity

Construction inventories (from a bill of material quantities, as far as possible; in some cases generalised across the catchments) for the distribution, collection and treatment systems were included in the LCA inventory data by Friederich *et al.* (2009b). Chemicals and electricity were included in the operational data inventories for water treatment and water recycling. Electricity only was included in the operational data inventory for wastewater treatment.

Friederich *et al.* (2009b) noted that the CML methodology is European-based and posed some limitations to their LCA application in the local (South African) context, namely:

1. Water as resource, salinization and loss of biodiversity (from land use) are very important locally but not included as impact categories; and

2. The way in which environmental impact category methodology (i.e. land use affecting ecotoxocity, acidification, eutrophication etc.) was developed and applied in the LCA models will be European-based and not necessarily appropriate for the local context.

Despite these limitations, some of the key points observed by Friederich *et al.* (2009b) have relevance to the AWRCOE project here. These are listed below:

- Electricity use (generated mainly from coal in South Africa) was the dominant contributor to environmental burden across all impact categories. Similar conclusions have been reached for water systems in Belgium and Italy, cited by Friederich (2009b).

- Chemicals use contributed (compared with electricity use) relatively less to the environmental burden across all impact categories (indicatively <1% to about 30%, typically 5 to 15% of the combined environmental scores for electricity plus chemicals)

- Secondary wastewater treatment (activated sludge system) followed by the sewerage collection (pumping) system were the two largest electricity users in the urban water system of eThekwini (greater Durban study area), This highlighted that greater extent of sewerage catchment covereage and higher levels of treatment for wastewater (a requirement for centralised water recycling) need to be considered for the system as a whole in debating the merits of water recycling. If the base case (without water recycling) involves a lower level of treatment (e.g. primary only, followed by ocean

discharge) using (potentially) a predominant gravity component in the collection system, then water recycling (with more pumping and more advanced treatment as a pre-requisite) will be less energy efficient and likely to have a greater environmental burden.

◗ Reducing consumer demand and system water losses are potentially very effective ways of reducing environmental burden for water systems, and should be considered first.

◗ When comparing recycling of wastewater (non-potable recycled water to industry) as an alternative to the equivalent provision of 'virgin' water (treatment and distribution of potable water), then recycling can potentially give a reduction in environmental burden of 68% to 88%, depending on the LCA impact category.

One of the conclusions reached by Friederich (2009b) was as follows:

"In the case study presented, recycling has positive environmental consequences and the replacement of virgin water with recycled water leads to high environmental savings and helps with the conservation of a vital resource. However, in the assessment of the environmental performance of recycling, an individual approach is needed since the environmental benefits can sometimes be smaller than the environmental burdens of the recycling process and the transport of the recycled water. In general…..recycling operations should be encouraged if there is demand for industrial-grade water in the vicinity of wastewater treatment plants. This process will be even more efficient in inland locations because the standards for discharge are higher for rivers than the sea and only a small improvement in the quality of the discharged water will bring it to industry standards. Recycling will become environmentally inefficient if the quality of the wastewater is very low, the quality of the recycled water is required to be very high and if significant pumping and energy are needed to transport the recycled water to customers."

3.8 Lundin *et al.* (2000, 2002) – Sweden

3.8.1 Lundin *et al.* (2000)

Lundin *et al.* (2000) used LCA methodology to compare the environmental loads from wastewater systems using different technical solutions. This study compared proposed conventional wastewater systems, both large and small scale, with separation systems: one in which urine is handled separately and one in which black water is treated in a liquid composting process. The study highlighted at least two important points that are indirectly relevant to the AWRCOE project here, namely, that:

◗ Large economies of scale, in environmental terms, can be gained both for the operation and construction phases of wastewater systems (i.e. compared to smaller decentralised plants, larger centralised plants tend to have lower environmental impact potentials through more efficient use of materials and energy). This is likely also to be true for water recycling plants.

◗ The urine separation systems outperformed the conventional systems by showing lower emissions to water and more efficient recycling of nutrients to agriculture, especially of nitrogen but also of phosphorus. This implies that the use of separation systems could significantly reduce the need for, and hence the production of, mineral fertilizers and thus reduce the overall use of energy and phosphate minerals. The combination of large-scale wastewater treatment and urine separation was found to be especially advantageous in these respects.

Lundin *et al.* (2000) concluded that some of the most important environmental advantages of separation systems emerge only when models of wastewater systems are expanded to also include potential effects of the production and use of synthetic fertilizers.

Whilst water recycling plants are not *ipso facto* designed to achieve nutrient removal, it is true that they are often indirectly responsible for driving higher levels of nutrient removal from wastewater. High product water quality requirements (removal of organics and pathogens, including viruses) for PRW plants (i.e. indirect potable reuse) is often a driver toward the inclusion of membrane processes in these plants, typically using the microfiltration-reverse osmosis process (MF-RO) sequence. Where reverse osmosis is employed to maximise product water recovery (e.g. >78% in three-stage RO), advanced pre-treatment including chemical coagulation with metal precipitants is often the preferred process design[7], which also gives advanced phosphorus removal to very low soluble concentrations (e.g. ≤0.1 mgP/L). Irrespective of advanced P removal through pre-treatment, limits are usually placed on the intake water quality to MF-RO plants in order to optimise PRW operation (e.g. minimise chemical cleaning requirements) and achieve product water quality control (i.e. for risk management reasons). These limits are applied to the secondary effluent from the wastewater treatment plant (WWTP) accepted as intake water to the advanced water treatment plant (AWTP) producing PRW and (indicatively) would include:

◗ Ammonia (<2 mgN/L) and turbidity (< 40 NTU) as a general indicator of activated sludge wastewater treatment plant 'health' or stable operation, and hence:

  – Log removal credits for pathogen removals;

  – Acceptable removal of organics to minimise fouling of PRW plant membrane processes;

◗ Soluble phosphorus[8] (<2 mgP/L) in order to minimise the risk of scaling in RO membranes and reduce chemical dosing requirements (as anti-scalants and/or for cleaning purposes).

It follows that PRW plants are generally associated with wastewater or pre-treatment processes that include at least high levels of phosphorus removal and often biological nutrient removal. The phosphorus is removed to sludge streams (generating biosolids), which can potentially be recycled to agriculture. This gives the potential for the biosolids product to offset the use of artificial fertilisers, based on its fertigation and soil enhancement properties.

From the findings of Lundin *et al.* (2000), biosolids and their potential for fertiliser offset, need to be included in the system boundary for an AWTP (PRW) plant LCA study. Irrespective of where it is physically achieved (i.e. at the WWTP or AWTP), where a portion of the nutrient removal (N and P) is attributable to the PRW plant operation, this needs to be accounted in the LCA model. For phosphorus it can be accounted for as biosolids, with a certain artificial fertilizer offset, making allowance for likely agricultural practice (e.g. application rate, nutrient availability, fertigation and crop yield value). For nitrogen and phosphorus removal (in the absence of urine separation processes for the catchment), this it can also be accounted for, through nitrification-denitrification and P-removal, as reduced nutrient emissions to the receiving water environment (i.e. lower eutrophication potential).

3.8.2   Lundin *et al.* (2002)

Lundin *et al.* (2002) noted in the introduction to their paper that the upgrading of wastewater treatment plants has historically evolved technology dedicated to "on end-of-pipe" technology where the main

---

[7] David Solley (Process Engineer, GHD), Personal communication (2012)

[8] Unless phosphorus reduction is provided in the membrane pre-treatment process

objective has been the efficient removal of environmental pollutants, particularly nutrients. They pointed out that:

"This reactive approach to environmental problems complicates the transition towards more sustainable urban water systems where the requirements for the reuse of water or plant nutrients are in focus… During the last decade, there has been an increasingly intensive desire to measure and describe different aspects of sustainability, with the focus often being on the environmental aspect." (Lundin *et al.,* 2002, p145)

Compared with other so-called Environmental Sustainability Indicators (ESI), Lundin *et al.* (2002) were of the view that LCA has the advantage that:

"… it is a well-established, standardised method which also includes an impact assessment phase (LCIA) where potential impacts are aggregated and quantified (ISO 14040, 1997; ISO 14042, 2000). LCA has been used for estimating environmental loads from urban water systems, usually wastewater systems (Lundin, et al., 2000). Selected LCA studies on urban water systems have revealed the <u>importance of nutrient recycling and energy recovery</u> (Tillman, Lundström, & Svingby, 1998; Lundin et al., 2000), which are often overlooked in the general discussion on the environmental sustainability of urban water systems. <u>A drawback with LCA is that it is a complex and time-consuming method</u>. There is a need for less complicated methods such as ESIs. Since one suggested application of LCA is to select environment performance indicators (ISO 14040, 1997), the basis for the development of ESI already partly exists. However, the ESI should not just measure environmental performance but need to operate on a <u>system wide level taking into consideration adjoining technical systems</u>….." (Lundin *et al.*, 2002, p146)

Lundin *et al.* (2002) presented an iterative procedure for ESI selection, using a framework based on LCA results and methodology, using case studies as an integrated part of ESI development. Their iterative procedure is shown in Figure 3-13. Some of the key points in the procedure were highlighted by Lundin *et al.* (2002) as follows:

"Once the overall purpose has been defined, the system boundaries must be defined (Lundin et al, 2000): "<u>Temporal, spatial and life cycle boundaries</u> must be addressed here. Since sustainability relates to prolonged time perspectives, temporal boundaries should be selected accordingly. A <u>time</u> (<u>temporal</u>) perspective of several decades is usually considered in the planning and construction of an urban water system. However, a longer time perspective of 50–100 years is required when the sustainability of urban water technology is considered.

In reality, <u>geographical</u> (<u>spatial</u>) boundaries for an urban water system are usually limited to include the municipality or watersheds, although the choice of life cycle boundaries has (recently) been demonstrated as a critical issue…..

The <u>life cycle</u> boundaries define the unit processes to be included in the system i.e. where up-stream and down-stream cut-offs are set. For the urban water system the life cycle starts with withdrawal of water from groundwater or surface water (Fig. 2) and includes drinking water and wastewater treatment. The life cycle ends with discharge of treated storm and wastewater to the aquatic ecosystem and incineration or disposal of sewage sludge, either to landfill or agricultural land. When such life cycle boundaries are used, these can also include the surrounding technical and agricultural system….."

The technical and agricultural system (or so-called 'technosphere') would include, for example, include production and supply of electricity, heat and fuels; transport and manufacture of chemicals; transport

and disposal of biosolids; manufacture, transport and use of articificial fertilisers, where these are offset by the agriculture disposal of biosolids).

The LCA framework includes all significant impacts on (or benefits to) the environment that takes place throughout the life cycle of the system defined, and relates these to a functional unit such as a person and year (Lundin et al, 2001). For water supply systems the functional unit could be a unit volume (e.g. 1 megalitre, ML) of water supplied.
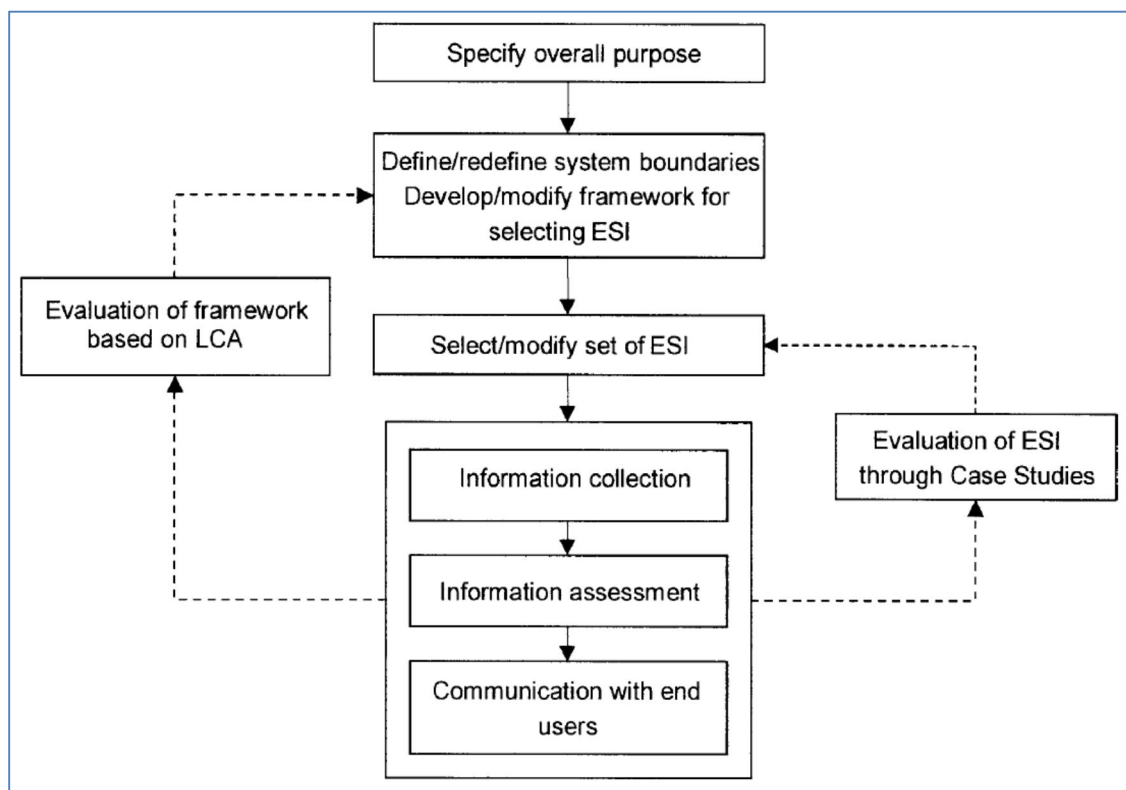
Figure 3-13 Interative procedure for assessing the environmental sustainability of an urban water system (reproduced from Lundin *et al.,* 2002)

In choosing ESIs, Lundin *et al.* (2002) recommended using the impact assessment step of LCA such that *"… as few indicators as necessary should be selected to address the important aspects"* for a given study. For system boundaries that are largely restricted to the ambit of influence of water utilities (i.e. the operation of the water/ wastewater treatment plants, the disposal route for biosolids and the number, type and source of chemicals used to treatment), a more limited set of ESIs was found to be adequate (Table 3-7). However, where the wider technosphere was included (e.g. including mining of raw materials for production of chemicals and electricity, agricultural fertilisers offsets etc.), then a wider set of ESIs was found to be necessary (Table 3-8).

Table 3-7   Environmental Sustainability Indicators (ESIs) selected by Lundin *et al.* (2002) for their case studies for urban water systems

Dimension	ESI
Withdrawal (raw water)	Annual freshwater withdrawal/ annual available volume
Water consumption	Use per capita per day (volume)
Treatment	Chemical and energy use for water supply
Distribution	Leakage (unaccounted water/ produced water)
Reuse of water	Reused water (volume)
Production	Wastewater production per day (volume)
Treatment performance	Removal of BOD_5, N & P
Loads of receiving water	Loads of BOD_5, N & P
Resource use	Chemical and energy use for wastewater treatment
Recycling of nutrients	Amount of N & P recycled
Quality of sludge	Cadmium content in sludge
Energy recovery	Energy recovered, heating and power

Table 3-8   Recommended ESI for assessing the environmental sustainability of urban water systems (extract from Lundin *et al.,* 2002, Table 3

ESI	Unit, all in $p^{-1}y^{-1}$	Relevance for environmental sustainability
Chemical use for drinking and waste water treatment	kg	Use of fossil energy and non-renewable resources
Electricity use for water supply	kWh	Contributes to use of fossil resources and related emissions
Electricity use for wastewater treatment	kWh	
Discharges of BOD, N and P to water	kg	Contributes to eutrophication
Discharges of selected heavy metals to water (Cd, Pb, Hg and Cu)	g	Contamination of aquatic ecosystems
Sludge to landfill	kg dry solids (DS)	Contamination of soil, water and air. Loss of potential useful resource
Total water use	m^3	Resource depletion. Influences the environmental impact over the entire life cycle
Use of hot water	kWh	Energy use and related resource use and emissions
Transportation of sludge for disposal	km	Diesel use and related emissions
Energy recovered from biogas	kWh	Avoids fossil resource use and related emissions
Energy recovered from heat pumps	kWh	Avoids fossil resource use and related emissions
Recycling of N to agricultural land	kg	Avoids fossil resource use and emissions of greenhouse gas (N_2O)
Recycling of P to agricultural land	kg	Avoids use of limited P resources.
Losses of NH_3	kg	Contributes to eutrophication and acidification.
Discharge of selected heavy metals to soils (Cd, Pb, Hg, Cu)	g	Contamination of terrestrial ecosystems

3.9   Godin, Bouchard Vanrolleghem (2012)

Godin *et al.* (2012) pointed out that the standard LCA approach in the past, when applied to wastewater treatment plants (WWTPs), was to focus on the main function of the WWTP as the removal of pollutants delivered to the plant in the form of incoming wastewater. This approach typically did not take into account the quality of the influent water, or the WWTP efficiency. Effluent and sludge quality (including heavy metal content) would, however, typically be taken into account. The result from such an approach is that environmental impact from the WWTP is, effectively, attributed to the WWTP, whereas in the urban water cycle, the pollutant load comes from the catchment and is not generated by the WWTP itself.

The problem here is one of boundary definition for the LCA studies of WWTP. The boundary needed to be extended to include the influent wastewater, treated effluent, biosolids stream and any gaseous emissions associated with the WWTP operations. Furthermore, the LCA results for the WWTP need to be compared to some reference case in order to assess the net environmental benefit of providing treatment. That is, a no-treatment scenario needs to be included as a reference case against which to assess the WWTP. A 'no treatment' scenario was included in the LCA approach taken by Lundie *et al.* (2004, see above) and by Foley *et al.* (2010) for wastewater treatment systems.

Godin *et al.* (2012) illustrated their work with a case study from WWTP in Quebec province (Canada). They introduced the useful term 'net environmental benefit' (NEB) to formalise the calculation of impact potential and expression of the results. NEB can be either positive (where the environment benefits overall from all the activities within the system boundary for a given category of impact potential) or negative (where, conversely, the environment is harmed overall). NEB was defined by the following simple equation by Godin *et al.* (2012):

$$NEB = IP_{NO} - IP_{TW} - IP_{SLC}$$

Where:

◗ IP denotes impact potential (for a given category such as global warming potential, eutrophication potential, ecotoxicity, metal depletion etc. generated in quantitative terms using LCA methodology).

◗ IP_{NO} is the impact potential associated with no treatment (i.e. the reference case in this study) in which untreated wastewater would hypothetically be discharged directly to the receiving water.

◗ IP_{TW} is the impact potential associated with the treated water (i.e. separating the impacts associated with discharging treated effluent to the receiving water environment from those associated with the WWTP itself).

◗ IP_{SLC} is the impact potential associated with life cycle of the WWTP including only those emissions or extraction of resources associated with the plant construction, operation, etc.

In the standard LCA approach, the avoided impact potential due to wastewater treatment ($IP_{NO} - IP_{TW}$) would typically not be considered and the treated water and WWTP life cycle impact potentials would be lumped ($IP_{TW} + IP_{SLC}$).

The study by Godin *et al.* (2012) highlights the importance of the reference case when interpreting LCA outputs. From the perspective of the AWRCOE project here, it will be important to establish which scenario(s) for alternative urban water supply will be used as the reference case(s) against which water recycling (e.g. indirect potable reuse) will be compared. For example, the relative change (or NEB) in

terms of global warming potential for water recycling might be positive (less impact) compared with seawater desalination, assuming electrical power comes from the grid for both; however, it might be negative compared with the conventional dam supply. A different conclusion would be reached for dam supply when considering the land occupation as the impact category. Moreover, the question of dam supply as a reference case might be controversial or ruled out on the basis of catchment or socio-political considerations. Hence, the reference case needs to be carefully selected and is likely to be project-specific.

4.   Reports and studies

Two published Australian studies and one international study were also identified as part of the literature review. The following section provides details on the objectives, methodologies and key outcomes.

4.1   Urban Water Security Research Alliance (Queensland) – Gold Coast Urban Water System Study (Lane *et al.,* 2011)

4.1.1   Summary of the Study

The Gold Coast region of south-east Queensland has a population of approximately 515,000 persons and is projected to increase by up to 16,000 persons per annum. This case study examined the urban water cycle of the Gold Coast under two scenarios of infrastructure provision, namely: (1) the 'Traditional' mix, as operating in 2007-8, prior to the commissioning of large-scale water recycling and desalination schemes in this region; and (2) a semi-hypothetical 'Future' mix that included seawater desalination and water recycling ('third pipe' Class A+ to a limited number (approximately 6%) of the total households; and indirect potable reuse via a dam from the largest wastewater treatment plant in the area). To some extent, the 'Future' mix options have already been implemented in the region. For the 'Future' infrastructure scenario, the population base was expanded by a large margin (2.6-fold) compared to the 'Traditional' scenario benchmark (2007-8 operating data). By assumption, the projected future water supply needs of the region would be met largely by a combination of desalination, water recycling and domestic rainwater tanks, but only limited additional supply from dams. Life Cycle Assessment (LCA) methodology was applied to these two scenarios in an effort to identify as broadly a possible the environmental impact potentials of adopting more energy and materials-intensive future water supply alternatives in order to service a growing population. The study found that the technologically more complex options of water supply (i.e. water recycling, desalination and even rainwater tanks) have the potential to increase environmental impacts on a 'per unit volume of water' basis in most impact categories (including ecotoxicity, human toxicity, global warming, ozone depletion and metals depletion), compared with the 'Traditional' supply via dams. Only eutrophication potential (through more advanced treatment and reduced nutrient load discharges) and freshwater extraction (through reduced reliance on dams) would be reduced, in relative terms. Achieving a balance between these opposing environmental impact potentials presents a challenge to urban water planners and operators in the context of achieving sustainability goals.

4.1.2   Study status and objectives

This study was commenced in 2007 (at the time when major desalination and water recycling projects had commenced in SE Queensland) and completed in 2011. The State Government also introduced new legislation in 2008 that encouraged the installation of rainwater tanks for new houses. Using the Gold Coast urban water system as an example, the objectives of this study were to investigate the following:

1. The greenhouse gas intensity profile of the system and its infrastructure components;

2. The spread of system impacts across a wider range of life cycle categories (not only global warming), including an improved understanding of the metrics required to estimate those impacts;

3. Life cycle impact trade-offs involved in choosing between alternative water supply options;

4. Key data gaps that should be priorities for further research; and

5. Overall assessment of the potential usefulness of LCA in planning decisions for such systems.

Water supply as well as wastewater collection and treatment were included in the study so as to study the whole water cycle. Impacts associated with each of the system components were assessed and compared, starting at the inventory level, using LCA methodology.

4.1.3   Methodology

Life Cycle models were set up for 'Traditional' and 'Future' scenarios. Data was collected and/or modelled for the construction and use of the infrastructure items included in each scenario. Construction inventories were extrapolated from other studies where local data was not readily available, and were annualized based on estimates of equipment lifespan.

Modelling of the operations phase captured all key operational inputs (e.g. chemicals and power) and outputs (e.g. disposal or reuse of wastewater and biosolids). The operations phase was based on a large set of actual inventory data collected as part of this study, with extrapolation from other studies or expert opinion to a limited extent. Due to space restrictions here, uncertainty estimation in the LCA model inputs and outputs are not considered in this review but were discussed in the report for this study (Lane *et al.,* 2011). Data for second order inventories (e.g. chemicals manufacture) were taken from available life cycle inventory databases available using the SimaproTM LCA software platform (PRé Consultants, The Netherlands). Australian database inventories were used as far as possible.

Impact assessment was performed using SimaproTM and the ReCiPe impact method (Goedkoop, Heijungs *et al.* 2009), which incorporates substantial improvements in terms of both fundamental impact science and LCIA methodology. The ReCiPe eutrophication impact method was adjusted to better reflect a mix of fresh and marine local receiving waterways.

Each so-called 'mid-point' indicator implies the potential for environmental impact, rather than attempting to predict actual environmental damage. Included were: Freshwater Extraction; Aquatic Eutrophication Potential; Ecotoxicity Potential (Marine, Freshwater & Terrestrial); Global Warming Potential; Ozone Depletion Potential; Fossil Fuel Depletion; and Human Toxicity Potential. Metals Depletion was included where relevant, mainly because of metals resource use in construction inventories, including pipelines. Despite the relevance of urban water systems to the global phosphorus balance, phosphorus resources are poorly represented in the available LCA impact models, and this currently represents a limitation.

4.1.4   Results

Breakdowns of the results for the 'Traditional' and 'Future' scenarios are summarized in Figure 4-1 and Figure 4-2 respectively. In these figures, the quantum of impact potential indicator per year (y) is given at the top of the column for each impact category. Note that the number of households served increased from 220,000 in the 'Traditional' scenario to 570,235 in the 'Future' scenario.

Figure 4-3 gives an example of the summary output from the LCA models, when comparing the two scenarios considered on a "per household" basis. Here the functional unit for the analysis was defined as "The provision of water supply and wastewater services, for 1 year, to an urban population for the Gold Coast region of SEQ". Since assumptions of system water supply and balance are embedded in this definition, linear extrapolation to other areas with different water balances is not possible.

Table 4-1 shows the relative impact potentials for alternative water supply options (on a specific basis i.e. per gigalitre of water supplied to the household) considered in the Gold Coast case study.

4.1.5 Conclusions from the study

Whilst this study was focused on Gold Coast infrastructure options, the results and conclusions are informative to a broader debate on urban wastewater and water supply options. The incorporation of non-traditional (dam-based) water supplies in response to growing populations is likely to mean that future pressures for environmental mitigation by the urban water sector will be spread across a wider range of issues than has traditionally been the case. Using LCA, the following conclusions can be drawn from this study:

- Power use is a major point of distinction between the four alternative water supply options considered in this study. Predictably, impacts related to power for the urban water cycle will increase substantially in the future as more energy-intensive water supply technologies are adopted. However, this study found that the LCA approach helped to compare alternatives such as rainwater tanks and water recycling with more traditional systems. The relativities in power consumption and associated impacts are not necessarily obvious when planning for urban systems as a whole.

- While power use is known to be the biggest indirect source of greenhouse gas emissions for an urban water system, this study included allowance for direct emissions (principally nitrous oxide and/or methane from sewers, wastewater treatment and dam storage systems). We found that the LCA approach helped to quantify the relative contribution from such possible 'fugitive' emissions to Global Warming Potential. This, in turn, may be useful in developing greenhouse mitigation strategies.

- The significance of a number of environmental issues identified in this study demonstrates that greenhouse gas emissions are not an adequate proxy for the range of important environmental externalities associated with urban water system operations.

- LCA provides a number of impact models that may enhance a broad spectrum environmental analysis of urban water systems. A number of areas were identified where current LCA methodology for urban water systems analysis could be improved. Notably, LCA models should be extended to consider the significance of phosphorus recovery in the context of global minerals resource depletion challenges.

- Similarly, regarding nutrient balances for any land application of biosolids and wastewater, existing LCA models suggest a potential for significant nutrient transfers to adjacent waterways. The likelihood and implications of achieving offsets from lower artificial fertilizer use also need to be considered. Quantifying fertilizer offsets and nutrient fluxes is subject to large uncertainties, and LCA in this area would be enhanced by guidance on best practice approaches to doing so in a regional context.

- Other than Human Toxicity Potential, the impacts considered in this study associated with infrastructure construction are likely to be of secondary concern from a life cycle perspective. This might not be intuitive to many people. It applied even for the relatively materials intensive rainwater tank and 'third pipe' water recycling reticulation systems.

- Wastewater treatment operations are the biggest source of most of the impacts considered, and may offer the greatest potential for reducing the overall environmental burden of the urban water system. Therefore, debates on the environmental implications of urban water system planning decisions need to have a wider focus than just the choice between water supply alternatives.

- Wastewater and biosolids pollutants (chlorine, metals & organics, including micropollutants) are the major source of potential ecotoxicity across the infrastructure lifecycle. Discerning the relative importance of these different contaminants is constrained by limitations with the available contaminant data, and the available LCA toxicity models. Dealing with the toxicity models required considerable research effort in this study. A number of areas requiring further research were noted.

- One of the benefits of the using the LCA approach is that indirect contributions to impact potentials can be tracked. Examples are transport (e.g. fuel use for biosolids and chemicals), mining and manufacturing (e.g. for treatment chemicals), which are responsible for substantial indirect contributions to the overall ecotoxicity potential associated with an urban water system.

- Where direct water recycling systems (i.e. 'third pipe' or similar) and household scale rainwater tanks are under consideration, quantitative comparisons of urban water supply alternatives (such as by LCA) should include sensitivity testing for different end-use demand levels and system configurations. The results in this study suggest that assumptions on the supply-demand balance for these systems could be critical to their ranking in comparative assessment of different water supply options.
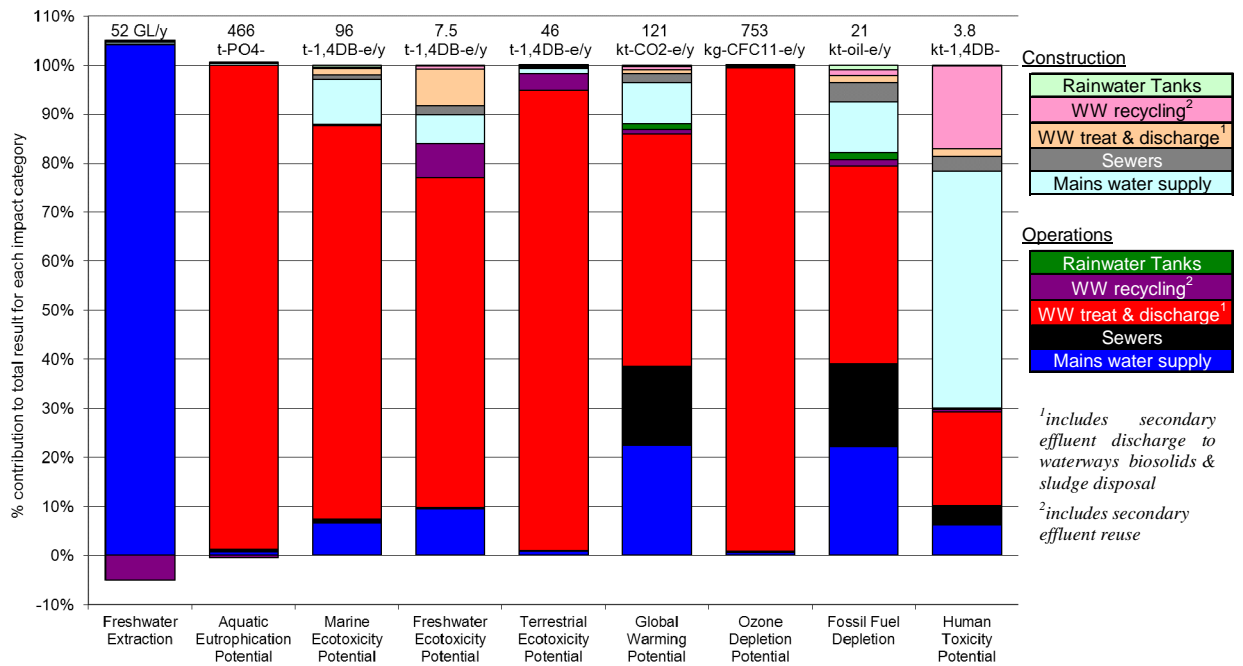
Figure 4-1 Breakdown of the LCA 'midpoint' impact potentials - 'Traditional infrastructure mix'
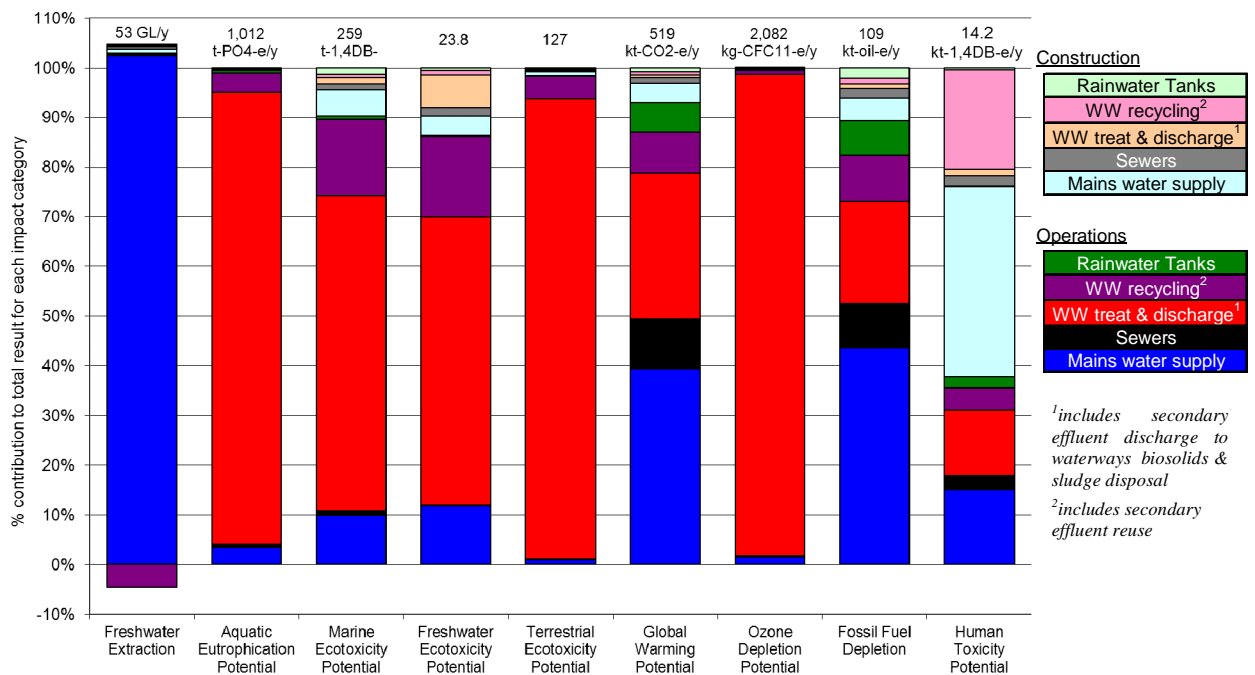
Figure 4-2 Breakdown of the LCA 'midpoint' impacts potentials - 'Future infrastructure mix'
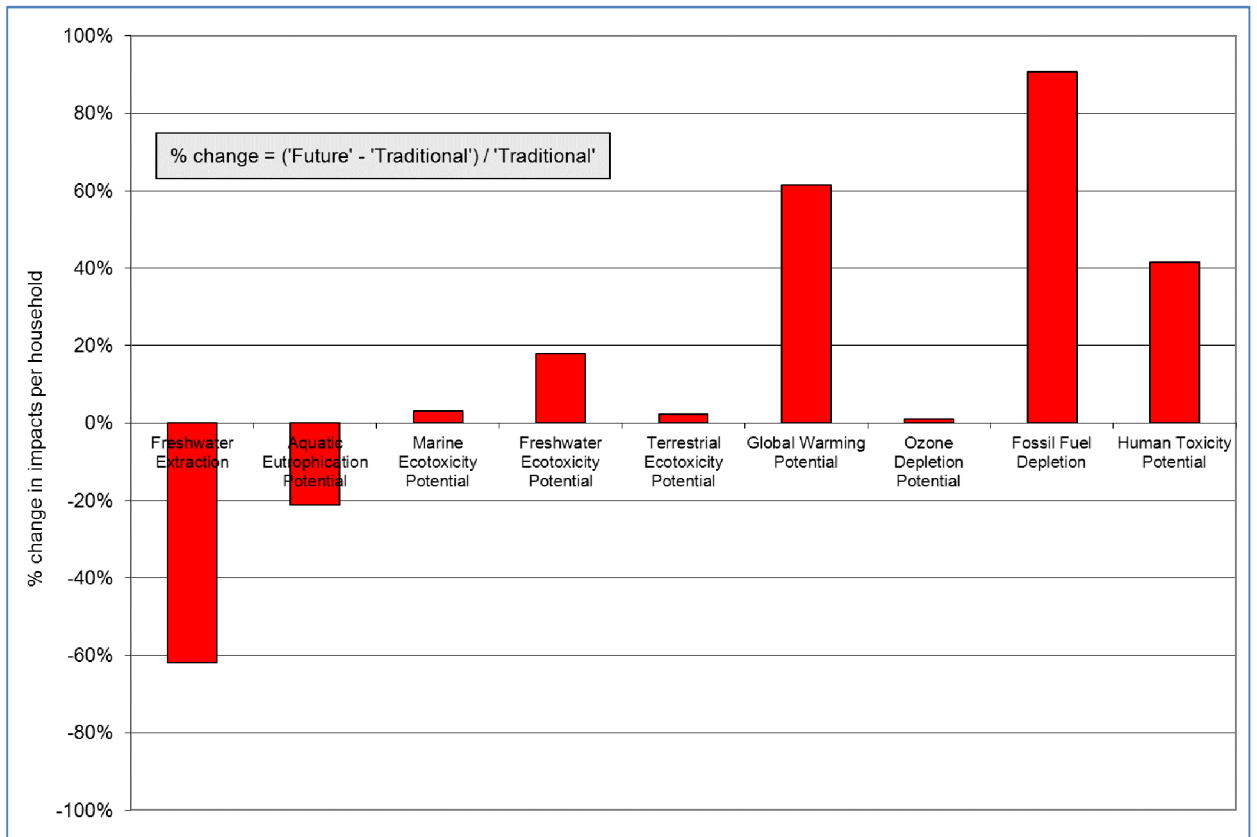
Figure 4-3   Relative change in midpoint LCA impact potentials moving from 'Traditional' to 'Future' scenario for the urban water supply system of the Gold Coast, including desalination and water recycling in the 'Future' scenario. Excludes wastewater collection and treatment here.

Acknowledgement: Figure 4-1, Figure 4-2 & Figure 4-3 from Lane et al. (2011) – Gold Coast Study

Table 4-1 Comparison of specific LCA impact potentials for alternative urban water supply options in the Gold Coast case study

Data from: de Haas, Lane and Lant (2011)

Water supply option:	Units of impact potential per GL mains water supply	Traditional Dam Supply	Class A+ (Pimpama-Coomera)	Class A+ (sidestream)	IPR (via Dam)	Raintanks (Low energy)	Raintanks (High energy)	SWRO Desalination
Impact category								
Freshwater Extraction	GL	1.067	0.004	0.002	0.008	0.003	0.012	0.010
Eutrophication Potential	tonnes PO4-e	0.1	-3.3	-1.6	-15.8	0.2	0.9	0.8
Marine Ecotoxicity Potential	tonnes 1,4DB-e	0.3	6.4	-1.4	-1.2	0.2	0.5	0.4
Freshwater Ecotoxicity Potential	tonnes 1,4DB-e	0.02	0.15	0.09	0.17	0.00	0.01	0.05
Terrestrial Ecotoxicity Potential	tonnes 1,4DB-e	0.02	0.49	0.25	0.03	0.01	0.03	0.02
Global Warming Potential	ktonnes CO2-e	0.725	1.245	0.578	2.395	0.933	5.318	4.415
Ozone Depletion Potential	kg CFC11-e	0.1	0.9	0.4	0.8	0.1	0.8	0.7
Fossil Fuel Depletion	tonne oil-e	135	291	136	584	273	1,369	1,078
Metals Depletion	tonne Fe-e	1.9	4.0	3.8	3.7	0.3	1.7	4.3
Human Toxicity Potential	tonnes 1,4DB-e	41	60	12	91	10	56	87

4.2 Yarra Valley Water Studies

Yarra Valley Water (YVW) the largest of Melbourne's three water corporations providing water supply and sewerage services to over 1.7 million people and over 50,000 businesses in the northern and eastern suburbs of Melbourne.

In ca. 2005, YVW embarked on a study entitled "The Sustainability of Alternative Water and Sewerage Servicing Options". The aim of the study was to explore the environmental impacts of providing different water and sewerage services to future 'green fields' (new) and infill developments. It was anticipated that the results from study will enable Yarra Valley Water to consider the environmental impacts of different options when undertaking strategic planning for water service provisions in upcoming new and infill developments.

Stage 1 study is summarised in a joint report by CSIRO Urban Water and RMIT (see Sharma *et al.,* 2005, in **References**) entitled: *Sustainability of Alternative Water and Sewerage Servicing Options*. The study focussed on a so-called "typical Yarra Valley Water Greenfield or Principal Activity Centre". The Greenfield site selected was Kalkallo and the Principal Activity Centre selected was Box Hill. Kalkallo was a proposed development for 86,000 people located approximately 30 kilometres north-west of Melbourne's CBD. Box Hill PAC, a central hub in Melbourne's eastern suburbs, is a high-rise development which will significantly increase the population density and demand on the water infrastructure.

Stage 2 of the study (GHD, 2009) involved a detailed analysis of the servicing options for Kalkallo urban water supply using an Integrated Water Management approach.

4.2.1 Kalkallo and Box Hill - Options Study, Stage 1

Approach

Since the project was aimed understanding the long-term sustainability of water servicing options, LCA methodology was used in order to evaluate not only 'present and local impacts' but also long-term global and 'inter-generational' impacts

The infrastructure conceptual design was aimed at quantifying various pipe materials used in water, wastewater and stormwater services. The data generated (for water and energy usage associated with the water cycle; wastewater & stormwater flows; contaminant loads; and various pipe materials required for the water and sewerage infrastructure) was used as input data for LCA.

In order to construct the conceptual design of water/ wastewater infrastructure, water balances and contaminant balances, the total residential, industrial and commercial areas were divided into various blocks. These blocks were further subdivided in to allotments. An average residential allotment of 533 m^2 was selected for planning purposes. Similarly an average industrial allotment of 1.5 ha and commercial allotment of 5 ha was considered. These allotments were further sub-divided into roof, paved and garden areas for the purposes of water balance analysis.

Water balance

Water balance analysis was conducted to identify the magnitude of water, wastewater and stormwater flows for various water servicing scenarios. These scenarios considered a range in levels of 'demand management' (i.e. water saving), as summarised in Table 4-2.

A summary of the water balances is given for Kalkallo and Box Hill respectively in Table 4-3 and Table 4-3 below. For further information on water and contaminant balances, pipework inventories etc. refer to Sharma *et al.* (2005).

Table 4-2   Water Supply Options considered in YVW Study (Sharma *et al.*, 2005)

Name Option	Demand Management	% Water Saving relative to Base Case
Kalkallo Development		
Option 1A - **Base case** to reflect conventional servicing option	Usual (UDM)	0%
Option 1B.1 - Precinct development to reflect recent practices (reclaimed water reuse) • Reclaimed Water for Toilet and Outdoor Use • Reclaimed Water for Toilet Only	'White Paper' ** (WPDM)	 43% 25%
Option 1B.2 - Precinct development to reflect recent practices (stormwater reuse)	'WPDM	43%
Option 1C – Self-contained development to reflect what can be done within building envelope (no reticulation)	High (HDM)	70%
Box Hill 'Principal Activity Centre'		
Option 2A - **Base case** to reflect existing conventional services	WPDM	0%
Option 2B - Local reuse within an individual building envelope (reclaimed water reuse)	HDM	53%
Option 2C - Precinct development of 20 buildings with centralised reclaimed water reuse ('sewer mining')	WPDM	38%

** 'White Paper' demand management refers to expected end for water adopted in the Victoria State Government White Paper – *Securing Our Water Future Together* (2004).

Table 4-3   Summary of average yearly water demand estimation for Kalkallo scenarios (Sharma *et al.,* 2005)

Scenario	Total Demand (ML/yr)	Demand met (ML/yr)	Imported Water Consumption (ML/yr)	Stormwater Runoff, not reused (ML/yr)	Wastewater to sewer, not reused (ML/yr)	Stormwater / Rainwater Consumption (ML/yr)	Greywater / Wastewater Reuse (ML/yr)
1A	14,439	14,439	14,439	9,731	11,420	0	0
1B.1	11,789	11,614	8,289	9,731	5,682	0	3,325 (RW)
1B.1Toilet	11,889	11,889	10,894	9,731	8,011	0	995 (RW)
1B.2	11,789	11,614	8,291	6,407	9,006	3,324 (SW)	0
1C	8,068	7,694	*4,108	7,301	5,422	2,335 (RT)	1,251 (GW)

SW-Stormwater, RT-Raintank water, GW-Greywater and RW-Reclaimed water

Table 4-4   Summary of average yearly water balance results for Box Hill scenarios (Sharma *et al.,* 2005)

Scenario	Total Demand (ML/yr)	Imported Water Consumption (ML/yr)	Stormwater Runoff (ML/yr)	Total Wastewater from PAC (ML/yr)	Wastewater Reuse (ML/yr)	Wastewater to Sewer, not Reused (ML/yr)
Option 2A	98	98	10	93	0	93
Option 2B	74	46	10	71	28	43
Option 2C	97	61	10	93	36	57

LCA system boundary

The system boundary considered by Sharma *et al.* (2005) for this study is summarised in Figure 4-4.

LCA outcomes - Kalkallo

For simplicity, Sharma *et al.* (2005) grouped the LCA impact category indicators into four groups:

- Global Warming
- Eutrophication
- Water use (i.e. Fresh water use from new supply)
- Solid Waste

Energy use and fossil fuel depletion therefore are indirectly reflected by the Global Warming (i.e. Greenhouse Gas) indicator. Other impact categories (e.g. photochemical emissions, carcinogens or other toxicants that might impact on ecosystems or human health) were not reflected in the results presented.

A summary of the LCA impact potentials, as grouped and presented by Sharma *et al.* (2005), is given in Table 4-5 for the Kalkallo development.
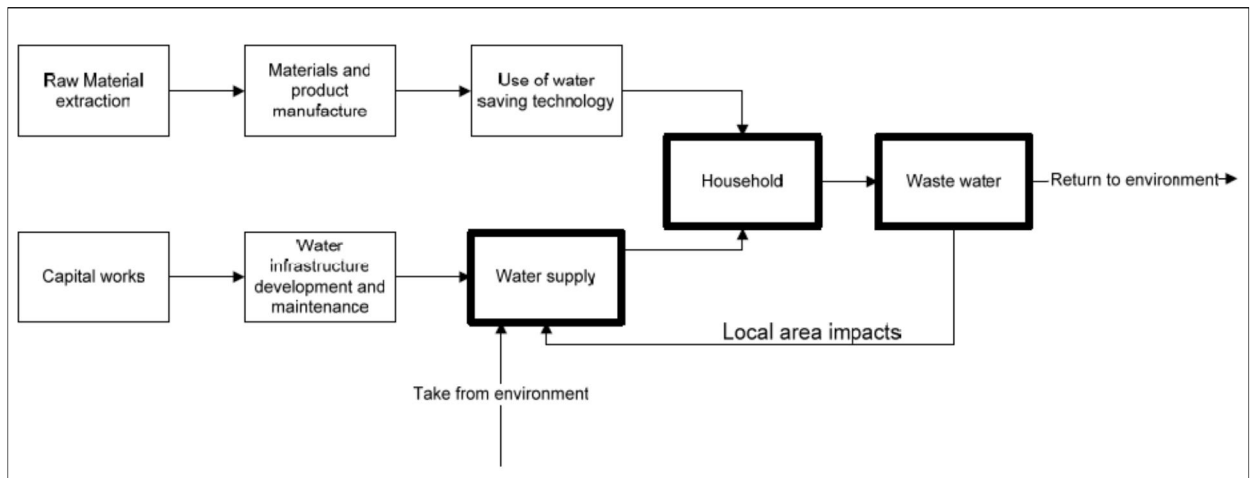
Figure 4-4   System boundary for LCA (Sharma *et al.,* 2005)

Table 4-5   Summary indicator results for the whole Kalkallo development per annum (Sharma *et al.,* 2005)

|	Global Warming	Eutrophication	Water Use	Solid waste
	kg CO2	kg PO4	KL H2O	kg
1A Conventional UDM	26290000	96570	14000000	5642000
1A Conventional WPDM	21520000	81050	11580000	5441000
1A Conventional HDM	15970000	65520	7994000	5208000
1B1 Dual retic. WW WPDM	21560000	60790	8084000	5342000
1B1 Dual retic. WW HDM	15980000	54540	6101000	5108000
1B2 Dual retic. SW WPDM	24220000	73860	8075000	5528000
1C Standalone HDM	17050000	43360	4277000	9379000

UDM = Usual Demand Management, WPD = White paper demand, HDM = Highest Demand Management

From Table 4-5 it is interesting to compare options that include wastewater reuse (i.e. water recycling via dual reticulation) against the conventional options (no water recycling) but the same modelled level of water demand management. For example, Options *1A Conventional WPDM* and *1B1 Dual retic. WPDM* have almost the same Global Warming potential (actually slightly higher for Option 1B1, with water recycling). The same applies when comparing Options *1A Conventional HDM* and *1B1 Dual retic. HDM.* Moreover, Option *1B2 SW WPDM* (i.e. Dual reticulation water recycling from Stormwater*)* has a 13% higher Global Warming potential than the equivalent conventional option (*1A Conventional WPDM*).

The shift in greenhouse gas emissions from reticulated water supply and reticulated sewage treatment and pumping (in the conventional options) to local wastewater treatment and pumping (for options with water recycling through dual reticulation) can be seen in Figure 4-5, when comparing equivalent levels of demand management (see above).

From these results it can be concluded that the main driver for water recycling is therefore not likely to be a reduction in greenhouse gas emissions.
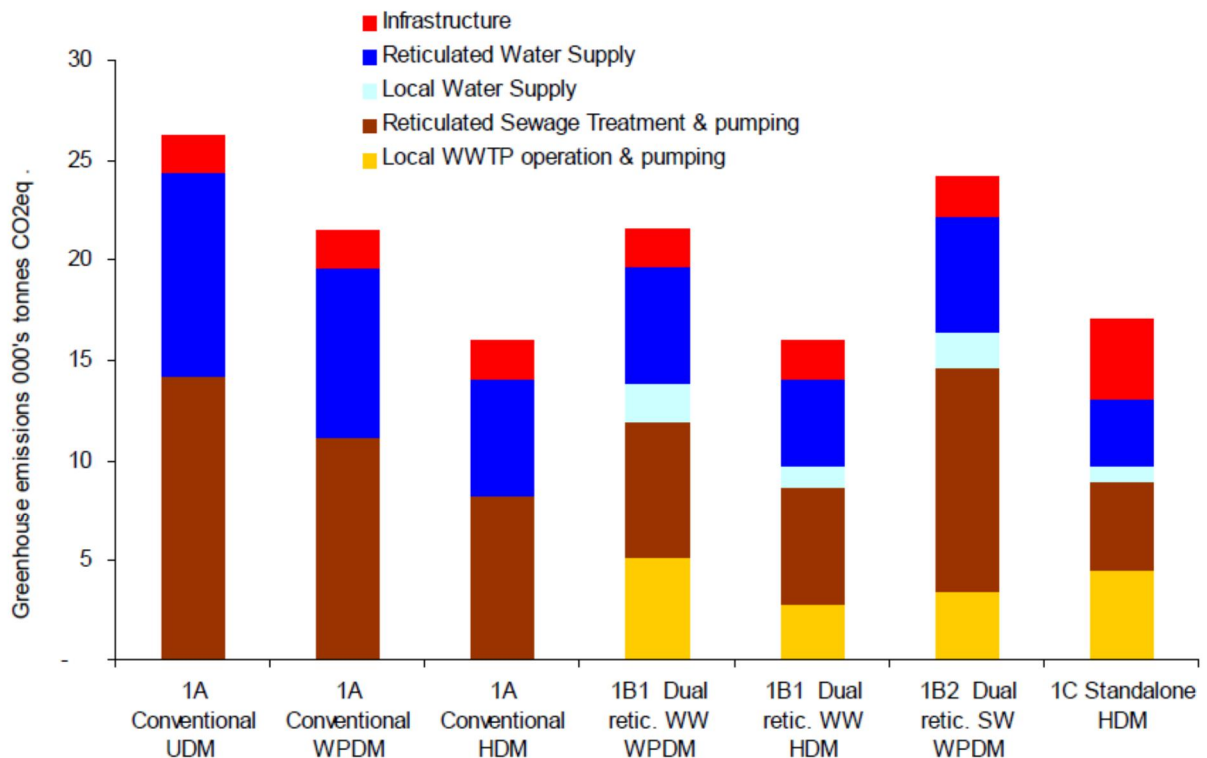
Figure 4-5   Greenhouse gas impacts by options for Kalkallo (Sharma *et al.,* 2005)

Table 4-5 shows that Water Use potential is lower (i.e less freshwater extraction from the natural environment) for all the options that include water recycling (as expected). From this it is clear that there is a trade-off: Global Warming Potential is unlikely to be lower for systems that include water recycling through dual reticulation but substantial Water Use savings are possible. In other words, as one would intuitively expect, water recycling saves freshwater supplies from the environment, but is not likely to reduce greenhouse gas emissions. Conversely, from the data presented by Sharma *et al.* (2005) due to lower water demand, greenhouse gas emissions intensity *per unit volume of water use* actually *increases* by between 44% to 61% for options with water recycling via dual reticulation and by 100% for the Standalone option of on-site treatment and disposal of wastewater effluent (refer to Table 4-5 above).

Sharma *et al.* (2005) also concluded that Eutrophication potential would decrease with increasing levels of demand management and water recycling. As they pointed out in discussion of these results, this conclusion needs to be carefully considered because it is predicated on two key assumptions, namely:

1. That the mass of nutrient removal from centralised (reticulated) wastewater treatment plants (WWTPs) will be proportional the volume of wastewater treated; i.e. that the concentration of nutrients in the treated effluent discharged will remain constant as the raw sewage concentration increases with decreasing sewage flow, due to lower domestic and commercial water use with higher levels of demand management. Implicit in this assumption is that WWTPs function more efficiently at higher raw sewage concentrations, within the bounds of the water end use models (commercially or domestically) adopted. There would be limits to the bounds within which this assumption might hold true and real data would need to be analysed to confirm its validity.

2. It was assumed that for the standalone treatment of wastewater on site (Option 1C in Table 4-5 or Figure 4-5) keeps nutrients on site and treats them in the aerated treatment plant with the effluent being disposed onsite for each house or other lot within the development. No nutrient escape was assumed from the site, which may be overly generous. The intention (according to Sharma *et al.*, 2005) was for this issue to be examined in more detail is Stage 2 of the study.

Table 4-5 shows that little change (2% reduction or less) in Solid Waste potential is brought about by including water recycling through dual reticulation, when comparing the same level of demand management. However, the Standalone option (Option 1C) with on-site treatment brings about a large *increase* in solid waste, due to the lack of economies of scale and shorter life expectancy for small on-site treatment plants and reticulation, as compared with centralised wastewater reticulation and large WWTPs.

One of the most striking observations from Table 4-5 (above) is that the biggest opportunities for reductions in impacts across all the impact categories considered is through higher levels of Demand Management. This is to some extent intuitive and similar to the findings of other LCA studies (e.g. Lundie *et al.* 2004, 2005 – refer to Section 3.1.1). **It highlights the fact that the biggest threat to sustainability is demand (consumption) itself, which is linked to growth in population numbers, human behaviour and the design of systems (e.g. urban water systems).**

LCA outcomes – Box Hill

The results for Box Hill (Table 4-6), suggest that there are better opportunities to reduce Water Use as well as reduce other impact potentials (Global Warming, Solids Waste and Eutrophication) where the development is more centralised or locally within high-rising buildings. Water Demand Management still offers the best opportunity for a reduction of impact potentials, but on-site dual reticulation of recycled water can offer further reductions in environmental impact (for example greenhouse gas emissions), particularly where the treatment takes place within the building envelope (e.g. Option 2B in this study - refer to Figure 4-6). For the case of dual reticulation of recycled water from sewer mining within a centralised precinct of several buildings, a lesser reduction in impact potentials may result, compared with the conventional option at the same level of demand management (Figure 4-6).

Table 4-6   Summary indicator results for the whole Box Hill development per annum (Sharma *et al.,* 2005)

Impact category	Unit	2A Conventional, WPDM	2A Conventional, HDM	2B Dual retic. Onsite WW, HDM	2B Dual retic. Onsite WW, WPDM	2C Dual retic. Sewer mine WW, WPDM
Global Warming	kg CO2	212200	165100	130700	170800	178100
Eutrophication	kg PO4	1734	1588	1411	1508	1511
Water Use	KL H2O	96760	74160	45580	60050	60160
Solid waste	kg	24810	22830	8384	10070	14150

WPDM = White paper demand, HDM High demand management
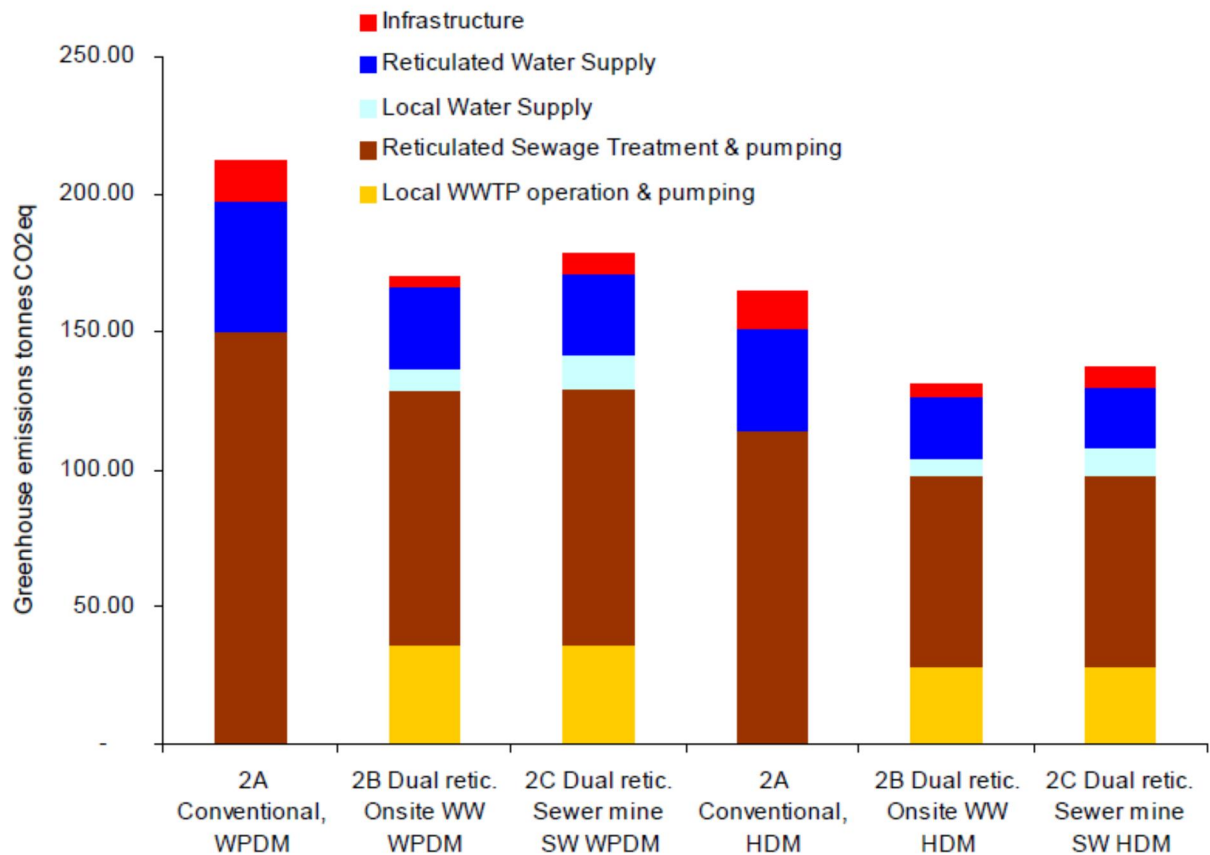
Figure 4-6 Greenhouse impacts by options for Box Hill (Sharma *et al.,* 2005)

4.2.2   Kalkallo and Box Hill - Options Study, Stage 2

This second stage of the study was conducted by GHD in conjuction with Life Cycle Strategies (GHD, 2009). The aim was to provide an integrated water management strategy that incorporates innovative integrated serving solutions for water systems, achieve environmental sustainability and provides a template for future planning. The modelling approach used is too detailed for review here but the key steps involved were:

◗ Integrated Water Management water balance modelling of the servicing options;

◗ Development of a concept design and layout of servicing infrastructure for each option (eleven options were considered in detail);

◗ Assessment of greenhouse gas footprint associated with each servicing option (using LCA methodology) – note that other life cycle impact categories were not used;

◗ Assessment of overall sustainability from an economic, social and environmental perspective using a Sustainability Assessment Framework (SAF).

The SAF is a form of multi-criteria analysis. It aims to consider a range of factors that typically influence decision making, broadly grouped into three main categories, or Primary Criteria (similar to 'triple bottom line' accounting):

◗ Financial (or Economic)

◗ Social

◗ Environment

A number of specific factors are then listed within each these three broad categories, named Secondary Criteria. Sometimes a further breakdown to lower-order (e.g. Tertiary) criteria within the Secondary Criteria is necessary. The SAF works by scoring each of the factors (typically at Secondary or Tertiary Criteria level), either from some measurable (or quantifiable) parameter (e.g. cost, volume or mass of discharge/ emissions etc.) or by some subjective score (e.g. in the case of social issues).

Figure 4-7 shows a breakdown of the Primary, Secondary and some Tertiary criteria used for the Kalkallo project (GHD, 2009), including the percentage weightings ascribed to each of these criteria. It is interesting to note that to a water utility (e.g. Yarra Valley Water in this case). sustainability is not only about evaluating impacts in terms of environmental (ecosystem), human health or resource depletion, as in Life Cycle Assessment methodology. Rather, sustainability is also viewed in Financial and Social terms, potentially traded off against Environmental impacts. Given that the Kalkallo study started with equal weighting to each to the Primary Criteria, it is revealing to note that one individual Environmental criterion at Secondary or Tertiary level (e.g. Greenhouse Gas Emissions; Mass of Wastewater N or P Discharged, i.e. linked to Eutrophication Potential) carries only a weighting of around 3 to 4% contribution to the total score.

The advantage of using an SAF approach is that it can be tailored to suit a particular project and includes factors (financial/ economic/ social) that might heavily influence a planning decision but are not captured in LCA. This suggests that LCA contribution information to the overall decision-making process for water-recycling or similar projects may be quite limited. LCA has the advantage of quantifying impact potentials over a broad range of categories, but once those impacts most relevant to a given project are selected and weighted into a framework that takes into account all other factors, the absolute contribution of the information provided by LCA to the final decision might be small. This calls into question the manpower requirements to assemble detailed life cycle inventories and building LCA models. For example, in the Kalkallo Study example, it was only the greenhouse gas emissions estimates from LCA in Stage 1 that ended up being included in the final SAF, for a weighting that counted just 3.3% to the overall score in Stage 2 (GHD, 2009).

Primary Criteria
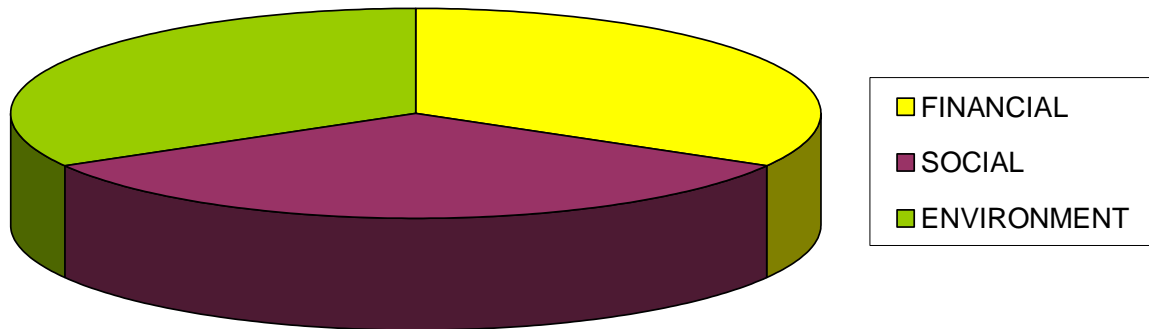
Secondary Criteria
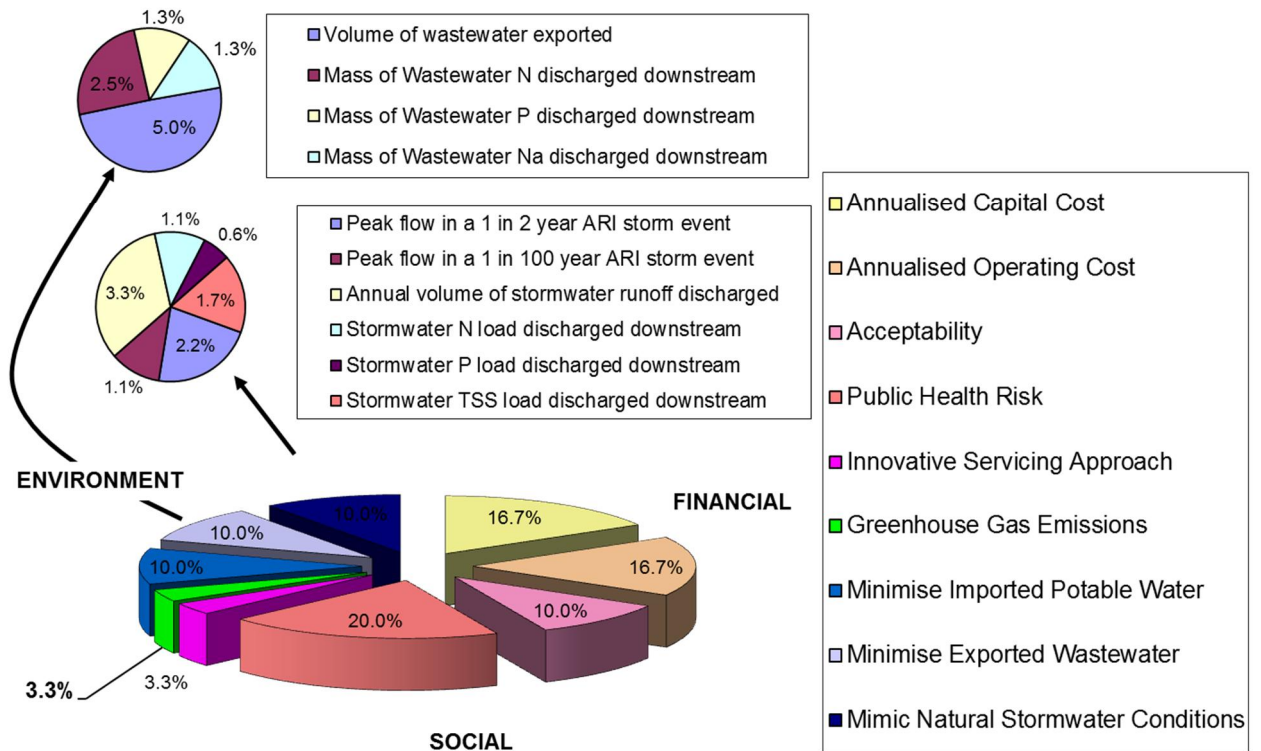
Figure 4-7 Breakdown of Sustainability Assessment Framework criteria & weightings used for Kalkallo (Melborne) in-fill development Integrated Water Management Options Study Stage 2 (GHD, 2009).

Note: Only Greenhouse Gas Emissions data came from LCA Study (Stage 1 of the project) for Kalkallo.

4.3 Orange County (California, USA) studies

Leslie (2007) and Leslie *et al.* (1999) summarised the value of water recycling as one of a suite of possibilities for supplementing water supply to growing cities. Leslie (2007) drew examples *inter alia* from Orange County in southern California (USA).

The Groundwater Replenishment (GWR) System in Orange County is scheme in which potable quality reuse water is produced from municipal wastewater at an advanced water treatment (AWT) facility located in Fountain Valley, California. Approximately 80% of the total water produced by the GWR System is piped to recharge facilities located near the City of Anaheim, while the balance is used to maintain and expand an existing seawater intrusion barrier.

The GWR system was built as a logical extension a 1970s reclamation project (named Water Factory 21), which has injected potable quality reclaimed water into the coastal aquifers to prevent ingress of seawater into the groundwater basin since 1976 (Leslie *et al.,* 1999). The GWR system is large, capable of delivering (by 2020) up to about[9] 123,400 ML/ annum (or about 338 ML/ day average) of recycled water. The drivers for the project included the following:

◗   Helping to prevent saltwater intrusion of groundwater aquifers used for water supply (a consequence of high groundwater extraction rates as a major source of regional water supply) – see above

◗   Improving regional water quality – Using reverse osmosis to produce lower salinity product water than that available from other water supply sources. By means of blending, the salinity (total dissolved salts or TDS) of urban water supplies in the region could be maintained at more acceptable levels, given the fact that high TDS water was known to be a major cause of corrosion of domestic and industrial pipework, heating and related infrastructure.

◗   Improving regional water supply security (southern California has faced a scarcity of water supply, relative to demand, for many years).

◗   Reducing reliance on water abstraction from river systems, including the State Water Project (from northern California) and the Colorado River, both of which had significant uncertainty around reliability and sustainability of supply.

◗   Reduced energy requirements associated with augmenting water supply, compared with other options (see above) that involved long-distance inter-basin transfers and very significant costs pumping energy costs.

◗   Improving regional wastewater management, through high levels of trade waste control (to limit toxins from entering the wastewater systems that are linked to the AWT facility producing recycled water) and reduced reliance on ocean outfalls for treated effluent disposal (with associated cost savings).

◗   Political benefits associated with aligning with the State of California Constitution, which states that "... the water resources of the State (must) be put to beneficial use to the fullest extent of which they are capable, and that the waste or unreasonable use or unreasonable method of use of water be prevented" (Leslie *et al.,* 1999).

---

[9] About two-thirds of the projected increase in water demand in southern California (150,000 acre.ft/ year) by 2020 (Leslie *et al.*, 1999). Note: 1 acre.ft = 1233.5 cubic meters = 1.2335 megalitres (ML)

From the above, it is clear that a number of wide-ranging drivers influenced the decision for the GWR system to go ahead in California. Some of these drivers were strategic, based on securing water supply and water quality for the region. Others had related environmental benefits (notably energy and reduced water abstraction from rivers or other catchments) but ultimately these drivers were largely strategic (i.e. energy cost savings; uncertainty of inter-catchment supply and high cost of transfer schemes; managing salinity etc.).

Leslie *et al.* (1999) and Leslie (2007) does not mention that life cycle assessment methodology was applied when the California GWR system was being planned. This might be partly due to the fact that LCA was a relatively undeveloped or emerging tool in the 1970s-90s. Nevertheless, some useful analogies can be drawn from the data represented by Leslie *et al.* (1999) and Leslie (2007) for the Orange County, California GWR scheme, which in many ways was a leader for its time.

Leslie *et al.* (1999) and Leslie (2007) drew attention to the relative energy requirements for water recycling. Some key observations from these papers can be summarised as follows:

◗ The energy requirements for water recycling using membrane treatment processes (e.g. microfiltration-reverse osmosis) are in the order of three to five times less than that required for seawater desalination.

◗ The energy requirements for water recycling in the GWR scheme (southern California - see above) are approximately 1.3 kWh per m^3 (or 1300 kWh/ML) (Leslie, 2007) to 1.84 kWh per m^3 (or 1840 kWh/ML) (Leslie *et al.,* 1999), depending on whether wastewater delivery and treatment plus recycled water conveyance are included. The breakdown is roughly as follows:

　　○ 0.2 kWh/ m^3 for wastewater delivery and treatment (assuming a 50/50 mix of treatment and ocean discharge)

　　○ (0.77 to) 1.11 kWh/ m^3 for GWR advanced treatment (including MF-RO)

　　○ 0.53 kWh/ m^3 for recycled water conveyance to GWR

◗ The alternatives to the GWR scheme for supplying treated water to southern California involved considerably higher energy requirements, namely:

　　○ Colorado River Aqueduct: 2.24 kWh/ m^3 (of which 2.0 kWh/ m^3 for delivery via pumping etc.)

　　○ State Water Project[10]: 3.50 kWh/ m^3 (of which 3.26 kWh/ m^3 for delivery via pumping etc.)

◗ The energy requirements for water recycling can appear relatively low (or attractive from a sustainability point of view) when the alternatives for meeting growth in water demand are higher (e.g. for southern California). However, in a different context, when compared for example against conventional dam supply options, the energy required for water recycling may be indicatively four times higher[11].

---

[10] Importing water from northern California

[11] Refer to data from de Haas *et al.* (2011) in Table 4-1 of this Review. Indirect Potable Reuse (IPR) fossil fuel depletion potential estimated to be 4.3 times higher than that for the Dams supply in the Gold Coast case study, on a 'per GL' basis for hypothetical alternative water supply options to the household, on a comparable system basis.

5. Conclusions

The objective of this review was to answer the question: *Can we distil sufficient useful information from previous LCA studies (published or otherwise, both in Australian or overseas) to demonstrate the use of LCA outputs as sustainability indicators in a NDEEP for reclaimed water? What are the knowledge gaps or inconsistencies?* Based on the literature reviewed, the following conclusions are drawn to address that question.

1. In order to rigorously address the question of sustainability around water supply and recycling, an LCA or similar study needs to include an entire urban water system within its boundary definition. For the NDEEP, the question of 'acceptability' will need to be framed within a defined urban water system context and scenario. Comparisons need to be made to some 'base case' (e.g. current scenario) for a defined context. The comparisons of estimated impact potentials or emissions from LCA can be made readily in relative terms for different options or scenarios.

2. Attempting to frame sustainability in general or in absolute terms (e.g. damage to the plant as a whole in terms of loss of human life, ecosystems or resources) is more complex and subjective. The context for sustainability then becomes damage due to societal impacts as a whole, which will be significantly more difficult for the NDEEP to address. Perhaps there is the possibility that the NDEEP from this project can in future form part of a larger program addressing larger questions around sustainability and growth for the nation as a whole, or indeed, globally.

3. A number of important assumptions need to be addressed when articulating sustainability within a defined urban water system context and scenario considered. Some examples of assumptions that might be tested are:

   – The urban water system is largely pre-existing (e.g. having evolved from urban infrastructure in Australia cities over the past 50+ years), as opposed to new urban developments with more modern, potentially water-sensitive designs.

   – Centralised treatment accounts for the majority of current urban wastewater systems and feasible options for use of recycled water by irrigation or other means, either: are limited on a volume basis; or have already been accounted for; or have already been implemented as far as practically possible.

   – Centralised treatment provides significant economies of scale and is a requisite for financially feasible large-scale water reclamation to augment drinking water supplies.

   – Water saving (e.g. 'demand management') measures are already in place, to the limit of practically or long-term public or industry acceptability and economic feasibility. Even with these measures in place, the total system water demand predicates that water recycling to augment drinking water supplies be examined as one option.

   – Energy saving measures are already in place, to the limit of practicality or long-term public or industry acceptability and economic feasibility. The energy inputs for water reclamation or recycling cannot be "offset" by saving energy elsewhere in the urban water system, or its associated activities.

4. Building an LCA model on the scale and complexity of an entire large urban water system is a resource-intensive process. It can only be justified for the NDEEP in this project if a case study and system boundary can be defined for which an LCA model is required to address significant

knowledge gaps in the existing literature. Some level of 'testing' of the value of indicative outputs from such an LCA model should be carried out with focus groups, or stakeholders, that might apply the NDEEP deliverables. This will help to determine to what extent LCA (or similar tools) might generate useful outputs to inform the NDEEP or enhance its value to stakeholders and the respective target audiences.

5. There is good existing data in the literature on average energy (and associated 'Scope 2' greenhouse gas) intensity for water supply to existing large urban water systems of major Australian cities. However, the data shows a wide range, reflecting regional differences in mix of supply (dams, groundwater, desalination etc.), topography, pumping distance and head, climate, and variable rainfall over time. Nevertheless to the extent that energy requirements tend to be the dominant inventory item in life cycle impact assessment for urban water systems, these data provide a useful benchmark against which the inclusion of water recycling can be compared as an alternative water supply source. Energy data for large and small urban recycled water systems are available in the data (e.g. Western Corridor Scheme in SE Queensland; case studies by Yarra Valley Water; Orange County in California).

6. Similarly, good data exists in the literature for the energy requirements of hot water heating or energy consumption broadly in the Australian economy. It may be informative to a debate on water recycling to consider the energy requirements, on a per capita population basis, against the energy consumption for such 'wider' benchmarks since the latter are indicatively one to three orders of magnitude greater.

7. The choice of treatment technology, scale (or system boundary) and type of water recycling can significantly affect its energy intensity and hence life cycle impact potential. For example, a water recycling project in South Africa using non-membrane technology (chemical coagulation/ settling/ dual media filters/ ozone/ granular activated carbon/ chlorination) to supply recycled water for nearby industrial use, was found to have a lower impact potential than the conventional potable water supply system for the location city, when compared on a 'per unit volume supply' basis. Yet comparing this recycled water application to the energy intensity of, for example, the purified recycled water via the Western Corridor Scheme in SE Queensland, then the latter is about eight times higher. Similar major differences appear when comparing the energy intensity of the large groundwater aquifer recharge scheme in Orange County (California, USA) with other potentially feasible alternatives (e.g. long-distance pumping or seawater desalination). These examples can serve to illustrate that the sustainability of water recycling schemes can be perceived as *relatively* better or worse, depending on the benchmark (and system boundary) against which the comparison is made.

8. Nutrient removal is an important consideration in the life cycle assessment of water recycling plants. Although not specifically designed for the purposes of removing dissolved nutrients (nitrogen and phosphorus) from water necessary pre-treatment, both upstream (typically in a wastewater treatment plant) and/or as part of the advanced processes required to produce purified recycled water for drinking purposes, results in advanced nutrient removal. How this is modelled (i.e. where the system boundary is drawn) can significantly influence the assessment of sustainability using LCA, but also adds complexity. For example, the reduced eutrophication through lower discharge of nutrients to receiving waters will show as ancillary benefits of water recycling. However, for completeness the disposal of sludge (or biosolids) also needs to be included in the models. Inclusion of sludge increases complexity due to uncertainties around: the downstream loss of nutrients and metals captured in the sludge through leaching into the terrestrial and aquatic environments; artificial

fertilizer offsets where sludge is disposed by beneficial reuse in agriculture; and fugitive gas emissions associated with sludge treatment processes and disposal.

9. Similarly, LCA models have the capability to calculate toxicity (terrestrial/ aquatic/ human) impact potentials. For example, inventory items like the generation of electrical power or transport using fossil fuels has significant potential indirect ecotoxicity burdens; or the material used for construction of additional pipelines for water recycling have significant indirect human toxicity potentials. However, there is some uncertainty over the model (characterisation and fate) factors that are implicitly part of how toxicity potentials are calculated in LCA methodology. Communicating uncertainty in the science and how this translates into LCA model outputs can be challenging, particularly to non-technical audiences, such in the NDEEP.

10. The extent to which such (above-mentioned) complexities associated with more rigorous LCA analysis can be effectively communicated to stakeholders and the target audience through the NDEEP is an important research question that needs to be addressed in this project. The associated research question is to what extent LCA findings are likely to influence perceptions of recycled water and its acceptability. In the absence of a case study specifically set up for the purposes of the NDEEP, there appear to be sufficient examples in the literature from which to draw examples of LCA outputs to test this research question.

11. Life Cycle Inventory (LCI) items (rather than calculated impact potential from life cycle models) may be simpler and easier to determine but equally effective as indicators of environmental sustainability. Examples of such inventory items related to water recycling might include: withdrawal (raw water); supply (chemical and energy use for water supply); loads of receiving water (e.g. nutrient and/or metal loads); or resource use (chemicals, materials and fossil fuel use for advanced treatment).

12. Recent examples can be found in the literature where LCA and sustainability considerations have been used as part of the evaluation of water recycling systems. Two case studies by Yarra Valley Water illustrate, like many others in the literature, that LCA is useful to the extent that it can give a quantitative assessment of impact potentials over a broad range of categories that can be grouped to represent ecosystems, human health and natural resources. However, it is interesting to note that for this project, Yarra Valley Water developed a decision making framework for assessing sustainability that takes into account only environmental but also social and financial factors. Within this framework, environmental sustainability was represented by: nutrient mass loads discharged to receiving water; abstraction of water from the available fresh water sources; and greenhouse gas emissions. Environmental sustainability was given a weighting of one third in the overall score, and within that, greenhouse gas received a weighting of only 3.3 %. Only the greenhouse gas impact potential was calculated using LCA methodology. This suggests that in real-life situations, there might be limited scope for LCA to influence the decision-making process around water recycling projects, given subjective weightings around competing factors (e.g. financial, socio-economic, or water resource considerations).

13. The literature on sustainability and application of LCA to water / wastewater systems does not consistently address the question of uncertainty of impact potential estimation. Uncertainty can arise from both activity data (i.e. inventory) and calculation of emissions estimates or impact potentials through impact models. This aspect will need to be addressed in this project in order to meaningfully compare different options or scenarios where estimates of sustainability impacts are inherently uncertain.

6. Recommendations

1. Stream 1 (and/ or Sub-stream 1.4) leaders should canvas their counterparts in Stream 2 and Stream 3 to gauge the type of output and appropriate level of detail suited to communicating sustainability messages associated with water recycling. There are sufficient examples in the literature from which to draw.

2. Any new case study modelled using Life Cycle Assessment as part of this study will require careful boundary definition in order to effectively capture (a) the base case against which water recycling for potable reuse is compared (e.g. conventional dam supply; desalination; pumped inter-basin water transfer); and (b) potential 'externalities'. Examples of such potential externalities are:

   – Indirect negative impacts arising from increased materials and energy use, as well as indirect environmental benefits from lower effluent nutrient and lower effluent metals loads caused by more advanced levels of pre-treatment (either at the wastewater treatment plant upstream of the advanced water treatment plant (AWTP) producing recycled water; or at the AWTP itself) biosolids

   – Indirect impacts (positive or negative) associated with biosolids disposal and fertiliser value (i.e. synthetic fertiliser offsets)

3. New LCA models (of a carefully defined case study) should only be built as part this project for the NDEEP if it can be shown that simpler indicators of environmental sustainability are inadequate for demonstration or educational purposes. Such indicators would typically be based on resource, materials and energy inventory items, for example relating to: the use of freshwater resources; energy consumption; use of materials; and discharge of nutrients/ metals/ other potential toxicants The advantage of rigorous application of LCA methodology is that it attempts to quantify impact potentials over a wide range of categories spanning the ecosystems health, human health and resource depletion as holistically as possible. The disadvantages with that such models are: inherently highly case-specific and tied to boundary definition; highly labour-intensive to build and interpret; subject to a significant degree of uncertainty stemming from characterisation and fate factors applied in the impact models within the calculations, e.g. for ecotoxicity impact potentials; and relatively complex to communicate to non-technical audiences.

4. The extent to which such the complexities associated with more rigorous LCA analysis, including uncertainty, can be effectively communicated to stakeholders and the target audience through the NDEEP is be an important research question that is to be addressed in this project.

7. References

Cook S, Hall M and Gregory A. (2012). *Energy use in the provision and consumption of urban water in Australia: an update*. Report prepared for Water Services Association of Australia by CSIRO (Water for a Healthy Country Flagship Report), May 2012.

DCCEE (2011). National Greenhouse and Energy Reporting System Measurement. Technical Guidelines for the Estimation of Greenhouse Gas Emissions for Facilities in Australia. Dept. of Climate Change and Energy Efficiency, Australian Government, Canberra, Australia.

De Haas D, Lane J and Lant P. (2011) Life Cycle Assessment of the Gold Coast Urban Water System. AWA *WATER* Journal (Dec. 2011), 38 (8), 57-64.

GHD (2009) Kalkallo Integrated Water Management - Analysis of Servicing Options. Report prepared for Yarra Valley Water, GHD Pty Ltd, Melbourne, Australia

Godin D, Bouchardt C and Vanrolleghem A. (2012) Net environmental benefit: introducing a new LCA approach on wastewater treatment systems. Water Sci. Technol. 65 (9), 1624 – 1631.

Hall M, West J, Lane J, de Haas D, and Sherman, B. (2009a) *Energy and Greenhouse Gas Emissions for the SEQ Water Strategy*. Technical Report No. 14, Urban Water Security Research Alliance, Brisbane, 2009. http://www.urbanwateralliance.org.au/

Hall MR, West J, Sherman B, Lane J and de Haas, D. (2009b) Long-term trends and opportunities for managing regional water supply and wastewater greenhouse gas emissions. *Environ. Sci. Technol.* 2011, 45, 5434–5440.

ISO 14040. (1997). Environmental management - life cycle assessment - principles and framework. International Organization for Standardization, Geneva, Switzerland.

ISO 14042. (2000). Environmental management- life cycle assessment, life cycle impact assessment. International Organization for Standardization, Geneva, Switzerland.

Jackson, T. (2009) *Prosperity Without Growth.* Earthscan, London, p67-86.

Jacob P and Whiteoak K. (2008) Energy Intensity of the Draft SEQ Water Strategy. Report prepared by Marsden Jacob Associates for Queensland Water Commission. Cited from Hall et al. (2009).

Kenway SJ, Priestley A, Cook S, Seo S, Inman M, Gregory A and Hall M. (2008) *Energy use in the provision and consumption of urban water in Australia and New Zealand*. Report prepared for Water Services of Australia (WSAA) by CSIRO (Water for Healthy Country National Research Flagship Report series ISSN 1835-095X), Dec. 2008.

Lane J, de Haas, D and Lant P. (2011). Life Cycle Assessment of the Gold Coast urban water system. Report submitted to Urban Water Security Research Alliance, Queensland. Technical report No. 52 Submitted Nov. 2011.

Leslie G. (2007) Pros and Cons of Water Recycling. *Precedent* 80 (May-Jun 2007), 30-34.

Leslie G, Mills WR, Dawes TM, Kennedy JC, McIntyre DF, and Anderson BP. (1999) Meeting the demand for potable water in Orange County in the 21st century: The role of membrane processes. *Proceedings of the American Water Works Assoc. Membrane Technology Conference*, Long Beach CA, USA.

Lundie S, Peters G, and Beavis P. (2004) Life Cycle Assessment for Sustainable Metropolitan Water Systems Planning. *Environ. Sci Technnol.* 38 (2004), 3465-3473.

Lundie S, Peters G, and Beavis P. (2005) Quantitative systems analysis as a strategic planning approach for metropolitan water service providers. *Water Sci. Technol.* 52 (9), 11-20.

Lundin M and Morrison GM. (2002). A life cycle assessment based procedure for development of environmental sustaintability indicators in urban water systems. *Urban Water* 4 (2002), 145-152.

Lundin M, Bengtsson M and Molander S. (2000). Life Cycle Assessment of Wastewater Systems: Influence of System Boundaries and Scale on Calculated Environmental Loads. *Environ. Sci. Technol.* 34, 180-186.

MFE (2011) Guidance for Voluntary Corporate Greenhouse Gas Reporting: Data and Methods for the 2010 Calendar Year. Ministry for the Environment, Wellington, New Zealand.

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.

Tillman A-M, Lundström H & Svingby M. (1998). Life cycle assessment of municipal waste water systems. *International Journalf LCA,* 3(3), 145–157.

Sharma A, Grant A, Gray S, and Mitchell G, Grant T and Opray L. (2005) Sustainability of Alternative Water and Sewerage Servicing Options. Joint report to Yarra Valley Water by CSIRO Urban Water and Centre for Design (RMIT University). Report available from Yarra Valley Water. http://www.yvw.com.au/yvw/groups/public/documents/document/yvw1001797.pdf

Sherman BS. (2001). Scoping options for mitigating cold water discharges from dams. p. 12–28. *In*: Phillips, B. [ed.] *Thermal pollution of the Murray-Darling Basin waterways. Workshop held at Lake Hume 18-19 June 2001. Statement and recommendations plus supporting papers.* Inland Rivers Network, World Wide Fund for Nature Australia.

Sherman BS, Ford P, Mitchell A and Hancock G. (2001). Greenhouse gas emissions from reservoirs - is Australian hydropower environmentally friendly? p. 1–10. *In:* Grimston, J. [ed.] Proc. NZSOLD/ANCOLD 2001 Conference on Dams: Dams – Development, Sustainability and Performance. The Australian National Committee on Large Dams (ANCOLD).

.

GHD

201 Charlotte Street Brisbane QLD 4000

GPO Box 668 Brisbane QLD 4001

T: (07) 3316 3000   F: (07) 3316 3333   E: bnemail@ghd.com.au

Document Status

Rev No.	Author	Reviewer		Approved for Issue		
		Name	Signature	Name	Signature	Date
0	D de Haas	P Ochre		P Ochre		
1	D de Haas	P Ochre		P Ochre		14.9.12
2	D de Haas	P Ochre		P Ochre		