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Life-cycle perspectives for urban water systems planning

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Abstract

Urban water systems around the world are going through a period of substantial change: they are evolving towards more complex water supply alternatives; are being placed under increasing pressure to achieve higher quality effluent and biosolids discharges; and are being confronted with a growing number of broader environmental management challenges. This thesis explored the use of the Life Cycle Assessment (LCA) methodology for assisting in that process of change, because of LCA's strengths in combining practicality and flexibility with 'big picture' thinking. The overarching goals were to: (1) explore LCA principles that do or do not provide useful perspectives for urban water planners, and (2) identify situations where the benefits from life-cycle thinking will be impeded by gaps in data and modelling approaches.

The analytical starting point was a comparison of two different configurations for a city-scale, integrated water supply and wastewater system: a 'traditional' approach dependent on low-energy dam-sourced mains water supply; and one with a more complex mix of contemporary water supply infrastructure intended to reduce the intensity of freshwater extraction and nutrient discharge. This change incurs a substantial increase in energy use, meaning the reduced pressure on local aquatic ecosystems may come at the expense of large increases in other life-cycle impacts. Notably however, the results generated in this thesis indicate that an exclusive focus on energy use is unlikely to be a robust approach to factoring these bigger picture environmental impacts into water industry decision making. Furthermore, it is the wastewater components of the system, rather than the water supply components, that make the largest contribution to most of the life-cycle impacts. An excessive focus on the energy or greenhouse gas (GHG) implications of growing urban water demands is, therefore, unlikely to chart the industry on an optimal course to a more environmentally benign system configuration.

The estimates for direct (scope 1) greenhouse gas emissions in this thesis utilise a comprehensive set of locally relevant empirical data and expert knowledge. Based on that, direct emissions could comprise 20% or more of the overall GHG footprint for urban water infrastructure systems. The substantial spatial variability associated with all the largest direct emission sources should be an important consideration in the urban water decision making process. For assessing the option to dispose of sewage treatment plant (STP) biosolids onto farmlands, the uncertainty associated with estimating field fluxes of carbon and nitrogen is likely to be more important than the more traditional focus on biosolids transport energy.

The second major case study considered in this thesis is focussed on the issue of biosolids reuse for agricultural purposes. When that practice is assessed against a broader set of impact categories than just energy use or GHG emissions, it becomes apparent that conventional life-cycle impact assessment (LCIA) models could bias against this as a

preferred fertiliser source. With respect to nutrient discharge, metals toxicity, and phosphorus recovery, there is a disconnect between the results produced with these impact assessment models, and the scientific knowledge and industry priorities that currently guide the associated Australian policy debate. Growing use of LCA in the Australian agricultural sector will encourage the use of those very models that are least well placed to provide useful critique of biosolids applications to soils, hence could lead to a weakening of agricultural support for this practice. This could pose a risk for water utilities already dependent on farmers to absorb the majority of their STP biosolids.

Phosphorus recovery and organics toxicity are both issues that could benefit from analysis incorporating the life-cycle perspective, since for both there is the prospect that water industry mitigation actions could shift the environmental burden to somewhere else in its supply chains. However, the analyses presented here suggest that the available LCIA models are not up to this task in either case. For the assessment of minerals resource depletion, the choice of impact assessment models could also have a substantial effect on the results that are obtained. A number of priority tasks are identified here, that would advance the LCA modelling framework so it can provide more meaningful contribution to urban water cycle planning.

Ozone depletion assessment is another issue where the adoption of conventional LCIA approaches will fail to provide any useful insight to the urban water industry. There is a strong case for including N_2O emissions in such assessments, and doing so clearly indicates this could have a material influence on the conclusions draw from analysis of water infrastructure systems. Quantifying the ozone enhancing effects of CO_2 and CH_4 emissions remains a bridge too far for the available LCIA models, but their increasing and complex influence suggests there may be a need for evolution in the metrics used to assess the ozone depletion issue. The urban water industry would likely be affected by any changes in international ozone-layer policy as a result of the increased scientific focus on these non-halocarbons, and should keep a watching brief on this issue.

The work undertaken in this thesis clearly identifies the value that can be derived from the LCA approach to infrastructure and options analysis. Furthermore, the compilation of whole-of-system data provides benchmarks that offer valuable benefits for the task of considering environmental trade-offs – whether that be from comparing across different water system technology options, and/or comparing across impacts that occur at different localities or points in time, and/or comparing across different environmental issues. In all respects, the goal should be to strive for robust consideration across all important life-cycle contributions and impacts. The challenge is to appropriately direct effort into the issues that matter, rather than those that are easiest to deal with. Incidental benefits derived from detailed industry data collection can be substantial, however they do not necessarily ensure that LCA delivers its

fullest value. Detailed consideration may well be required on uncertain issues that are beyond the traditional expertise of the analyst. Practitioners should also resist the temptation to uncritically adopt historical convention in the choice of LCA impact assessment models. Lack of rigour in the definition of inventory data and/or impact assessment models doesn't prevent LCA studies from being completed, but it will greatly diminish the value of the LCA exercise.

Declaration by author

This thesis is composed of my original work, and contains no material previously published or written by another person except where due reference has been made in the text. I have clearly stated the contribution by others to jointly-authored works that I have included in my thesis.

I have clearly stated the contribution of others to my thesis as a whole, including statistical assistance, survey design, data analysis, significant technical procedures, professional editorial advice, and any other original research work used or reported in my thesis. The content of my thesis is the result of work I have carried out since the commencement of my research higher degree candidature and does not include a substantial part of work that has been submitted to qualify for the award of any other degree or diploma in any university or other tertiary institution. I have clearly stated which parts of my thesis, if any, have been submitted to qualify for another award.

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Publications during candidature

Journal papers - peer-reviewed

Lane, J.L., de Haas, D.W., Lant, P. (under review) The diverse environmental burden of city-scale urban water systems. *Submitted to Water Research*

Lenzen, M., Geschke, A., Wiedmann, T., **Lane, J.,** *et al* (2014) Compiling and using input-output frameworks through collaborative virtual laboratories. *Science of the Total Environment* **485-486**(1): 241-251.

Lane, J., Lant, P. (2012) Including N2O in ozone depletion models for LCA. *The International Journal of Life Cycle Assessment* **17**, 252-257.

Hall, M.R., West, J., Sherman, B., **Lane, J.,** de Haas, D. (2011) Long-Term Trends and Opportunities for Managing Regional Water Supply and Wastewater Greenhouse Gas Emissions. Environmental Science & Technology 45, 5434-5440.

Book chapters

Lane, J. (in press) Stratospheric Ozone Depletion. Chapter 5 in 'Compendium of Life Cycle Assessment'. (Eds W Kloepffer and M Curran) Vol.4, Life Cycle Impact Assessment - LCIA. Springer.

Journal papers – not peer-reviewed

de Haas, D., **Lane, J.,** Lant, P. (2011) Life cycle assessment of the Gold Coast urban water system. *Water* **38**, 57-64.

Journal Papers – in preparation

Lane, J.L., Lant, P. (in prep) Biosolids and agriculture – a growing 'LCA risk' for urban water utilities.

Bradford-Hartke, Z., **Lane, J.L.,** Lant, P., Leslie, G. (in prep) Environmental benefits and burdens of phosphorus recovery from wastewater in Australia.

Malik, A., Lenzen, M., **Lane, J.L.,** Ralph, P., Tamburic, B. (in prep) Hybrid life cycle assessment of an algal biofuel industry.

Technical Reports

Lane, J., Binks, A., Kenway, S. (2013) 'Greenhouse Gas footprints – Seqwater infrastructure, and the broader urban water system', University of Queensland.

Lane, J., Lant, P. (2012) 'Application of Life Cycle Assessment to wastewater systems planning.' Urban Water Security Research Alliance, Technical Report No.96, Brisbane.

Lane, J., De Haas D, Lant, P. (2012) 'Life Cycle Assessment perspectives on wastewater recycling.' Urban Water Security Research Alliance, Technical Report No. 86, Brisbane.

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

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Conference Papers

de Haas, D., **Lane, J.**, Lant, P. (2012) *Life Cycle Assessment of an Urban Water System On the East Coast of Australia*, Proceedings of WEFTEC2012, USA, October 2012.

Lane, J., Lant, P. (2012) *Life Cycle perspectives for Total Water Cycle Planning*, UWSRA Science Forum, Brisbane, June 2012.

Lane, J., de Haas, D., Lant, P. (2010) *Life Cycle Analysis of the Gold Coast Urban Water System*, UWSRA Science Stakeholder Forum #2, Brisbane, September 2010.

Conference Abstracts

Lane, J, (2013) *Life-cycle toxicity modelling in the Australian context*, Poster presentation to ALCAS Conference, Sydney, July 2013

Lane, J, Wiedmann T, Lenzen M, *et al* (2013) *New capability for hybrid input-output life-cycle assessment – the Industrial Ecology Virtual Laboratory*, Platform presentation to ALCAS Conference, Sydney, July 2013

Lane, J. (2013) *Advancing Sustainability Science using a Virtual IO Laboratory*, Platform presentation to International Input-Output Association Conference, Japan, July 2013

Lane, J., Lant, P. (2012) *N_2O and Ozone Layer Depletion – A New Consideration for Urban Water Planners…?*, Poster presentation to the UWSRA Science Forum, Brisbane, June 2012.

Lane, J., Lant, P. (2012) *Beneficial use of biosolids – a Life Cycle Assessment perspective*, Platform presentation to the AWA Biosolids & Source Management Conference, Gold Coast, June 2012.

Lane, J. (2012) *Wastewater Recycling – the Life Cycle Assessment (LCA) Perspective*, Presentation to the special workshop on 'Using Ozone & Biofiltration for Producing Recycled Water', Ozwater, Sydney, May 2012.

Lane, J., de Haas, D., Lant, P. (2011) *LCIA – relevance & challenges for the urban water sector*, Platform presentation to ALCAS Conference, Melbourne, March 2011

Lane, J., Grant, T., Lant, P. (2011) *Normalisation for Australian LCA – improving the perspectives*, Poster presentation to ALCAS Conference, Melbourne, March 2011

Publications included in this thesis

1. **Lane, J.L.**, de Haas, D.W., Lant, P. (under review) The diverse environmental burden of city-scale urban water systems. Submitted to *Water Research*

The paper is wholly attached to Appendix B1. The supplementary information from the paper is wholly included in Appendix B2.

- Excerpts from the *Introduction* of this paper are modified and included in Chapter 2 (Motivation & Objectives) in this thesis.
- Excerpts from the *Methodology* and *Supplementary Information* are included in Chapter 3 (Research Methods) and Chapter 4 (Literature Review) in this thesis.
- The *Results* and *Discussion* are included in Chapter 5 (Research Outcomes) in this thesis.

De Haas and Lant initiated the study, which was then scoped by Lane, de Haas and Lant. De Haas collected the majority of the infrastructure operations data, and provided initial processing of that data. Lane collected all other infrastructure operations data, undertook the literature reviews to generate the remaining input assumptions, and produced the water balances that define the scenarios. Lane undertook the literature reviews to choose and develop the impact assessment models. Lane compiled all data, developed the system models, and generated all results. Lane, de Haas and Lant analysed the results. Lane scoped and drafted the manuscript. Lant and de Haas edited the manuscript.

2. **Lane, J.** (in press) Stratospheric Ozone Depletion. Chapter 5 in 'Compendium of Life Cycle Assessment'. (**Eds** W Kloepffer and M Curran) Vol.4, Life Cycle Impact Assessment - LCIA. Springer.

This book chapter is wholly attached to Appendix A1.

- Excerpts are modified and included in Chapter 2 (Motivation and Objectives) and Chapter 4 (Literature Review).

Lane undertook the literature reviews. Lane scoped, drafted and edited the manuscript.

3. **Lane, J.,** Lant, P. (2012) **Including** N2O in ozone depletion models for LCA. *The International Journal of Life Cycle Assessment* **17**, 252-257.

The paper is wholly attached to Appendix A2.

- Excerpts from this paper are modified and included in Chapter 2 (Motivation & Objectives) in this thesis. Excerpts from the *Results* and *Discussion* of this paper are modified and included in Chapter 5 (Research Outcomes) of this thesis.

Lane scoped the paper, undertook the literature reviews, compiled the case study and produced the analysis. Lane drafted the manuscript. Lant edited the manuscript.

Contributions by others to the thesis

David de Haas and Paul Lant initiated the project, and developed the initial concept of analysing the Gold Coast urban water system. David de Haas led the initial scoping of the Gold Coast system modelling, then collected the operations and infrastructure data for the Gold Coast Water mains water system and sewage treatment system. David de Haas compiled, analysed and processed that data, with support from Kelly O'Halloran and Mark Wilson of Gold Coast Water. The outcomes from that data processing were used as inputs to the case study models.

The following people provided guidance on developing other inputs to the modelling:

- Cara Beal (Griffith University) and Rachelle Willis (Griffith University) provided guidance on the residential water use assumptions that underpin the Gold Coast case study water balances.
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- Marguerite Renouf (UQ) and Damien Batstone (UQ) on the modelling of biosolids application to soils. Zenah Bradford-Hartke (UNSW) also provided input on the modelling of biosolids analysis, through the compilation of scenarios for her own PhD at the University of New South Wales.
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David de Haas and Paul Lant provided critical review of manuscripts included in this thesis.

Statement of parts of the thesis submitted to qualify for the award of another degree

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Keywords

life cycle assessment, lca, life cycle impact assessment, urban water, water supply, wastewater, biosolids, environmental impact assessment, ozone depletion, greenhouse gas

Australian and New Zealand Standard Research Classifications (ANZSRC)

ANZSRC code: 050204, Environmental Impact Assessment, 80%

ANZSRC code: 090702, Environmental Engineering Modelling, 20%

Fields of Research (FoR) Classification

FoR code: 0502, Environmental Science and Management, 80%

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1. INTRODUCTION

Urban water systems around the world are going through a period of substantial change (Beck 2011, see, for example, Brown et al. 2009, Marlow et al. 2013, Prosser 2011). To reduce their dependence on extraction from natural freshwater systems, many cities are exploring alternative approaches to water supply. Wastewater systems are also increasing in complexity, driven by the push for greater levels of nutrient removal, and stricter constraints on the disposal of biosolids. Understandably, local environmental concerns (ecological pressure in the source and receiving waters) and management concerns (maintaining a reliable water supply; finding somewhere to put the wastes) have been, and still are, major drivers for these trends.

The genesis for this project lies in the somewhat simple question – moving beyond these local concerns, what are the bigger picture environmental sustainability issues associated with these developments in the urban water sector? Improved understanding of such bigger picture concerns will allow the industry to better anticipate and plan for future changes in the social expectations and government policy that so strongly influence their operating paradigm.

This project aims to help with that task. In particular, it explores the use of the Life Cycle Assessment (LCA) methodology for developing an improved understanding of integrated urban water systems. While LCA has a long history of application to urban water systems analysis, section 2 outlines many gaps in that body of work that limit the usefulness of past and future LCA studies for the industry. This thesis aims to address a number of those gaps.

> ***PhD goal:***
> *Use Life Cycle Assessment principles to provide perspectives of relevance to urban water systems planning.*

2. MOTIVATION AND OBJECTIVES

Why use LCA ?

Of the many tools, concepts and bottom-up decision making systems that have been canvassed in academic and industry fora, LCA is one of the better options at combining practicality with big picture thinking.

The 'practicality' of LCA stems from its suitability for decision making in the everyday 'bottom-up' sense, and its methodological flexibility. The 'big picture thinking' aspect of LCA is not so much about analysing why things are the way they are, or why things should be different. Rather, it is about providing a framework for considering impacts that might occur beyond the remit of the decision makers' organisation and/or beyond the remit of the decision maker's location (Bauman and Tillman 2004). This is achieved through a combination of: (a) extending the impact analysis across the life-cycle, such that a comprehensive spatial and temporal breadth is included; and (b) extending the impact analysis across a range of environmental issues, providing comprehensive coverage of those issues relevant at all stages of the life-cycle.

To accommodate the complexity that can be introduced by such an approach, the LCA methodology aspires to a strictly quantitative assessment, with a high priority placed on rigorous definition of the analytical system boundary and the data that is included. This rigour is strongly backed by ISO standards that guide the implementation of LCA principles (Finkbeiner 2013, ISO 2006a, b, 2013, 2014). Collectively, these aspects of the LCA methodology are intended to help reduce the risk that something important gets overlooked in any particular options comparison, or any particular decision making context.

Even when such analysis can be delivered successfully, it is not a given that the results of an LCA study will, or should, change the decision making outcome. Life cycle environmental impacts will never be more than one subset of the issues being weighed up in a comparative options study. And LCA should not be considered any different from other multi-criteria analytical approaches – these deliver their greatest value when used as a tool for investigating the quality of data, assumptions and priorities; rather than in the expectation they will deliver 'magic' answers (Baitz et al. 2013, Kain and Soderberg 2008). This notion is elegantly captured in the sentiment that "the beauty of LCA is not necessarily in its objectivity. Certainly, LCA cannot make the choice for us. Rather it lays out all the assumptions and quantifies the estimates in the common denomination of dollars (currency), greenhouse gases, energy, materials and/or environmental impact for all to examine. It opens our environmental books in a transparent fashion, and that's its real strength." (Schnoor 2009).

An important consideration for the urban water industry, as for others, is the impressive momentum towards greater use of LCA across a range of domains (Chen et al. 2014, Peters 2009). Publishing trends in the journal *Environmental Science & Technology* are one such case in point - the history of this prestigious international journal is steeped in a focus on technical science, however the journal editors also identify a need to bridge the gap between science and policy. Their growing rate of LCA publications indicates the editors see LCA as an increasingly important tool in building that bridge (Guinee et al. 2011, Schnoor 2009). Research activity is only one part of the story, since the majority of LCA application in practice occurs through unpublished analysis by businesses (Baitz et al. 2013). This has moved beyond the realms of internal process assessment, with large retailers in the USA now requesting LCA data to support their supply chain choices (Fava et al. 2009). That business interest will, in part, reflect a consumer driven push for LCA information via both formal product labelling schemes, and the periodic but intense scrutiny that falls on particular issues such as life-cycle carbon or water footprints.

Finally, LCA is also being pushed from the top down. The policy momentum of LCA is perhaps most directly apparent in the European Union, where it has been directly incorporated into policies on product development and waste management (Wolf et al. 2012). LCA, or life cycle thinking, also appears in policies of other parts of the world, most notably in any countries that have 'green purchasing' requirements for government departments, or standards to encourage consistency in carbon footprint labelling (Wolf et al. 2012). The United Nations Environment Program is also a supporter, with a long standing interest in encouraging advances and standardisation in the LCA methodological research community (de Haes et al. 2002, Jolliet et al. 2005, Rack et al. 2013, Valdivia et al. 2013).

Why investigate the overall urban water system ?

In the previous section, emphasis is placed on the driver for LCA being its contribution to the bottom-up decision making process. This begs the question – why does this project use LCA to consider the overall urban water system, rather than focussing on specific components (e.g. water supply; wastewater treatment) of that system?

It is at the component level that the bottom-up decision making normally happens. This is reflected in the focus of most past, and contemporary, LCA analysis for the urban water industry. Of those studies available in the literature, a great many have compared choices to do with wastewater treatment or the handling of sewage sludge (see the literature surveys in Corominas et al. 2013, Foley et al. 2010a, Remy and Jekel 2012, Rodriguez-Garcia et al. 2011, Yoshida et al. 2013). Other studies, albeit far fewer, have compared or critiqued different technologies for wastewater recycling (e.g.

Munoz et al. 2009, Pasqualino et al. 2011, Tangsubkul et al. 2005). Many others have considered options for conventional drinking water treatment systems (see Bonton et al. 2012, Friedrich 2002, Igos et al. 2013, Vince et al. 2008) and alternative technologies such as seawater desalination or urban rainwater capture (e.g. Angrill et al. 2012, Del Borghi et al. 2013, Godskesen et al. 2013, Hancock et al. 2012, Mithraratne and Vale 2007, Muñoz and Fernandez-Alba 2008).

In contrast, this PhD started with a focus on overall urban water systems. Such analysis can provide useful perspective to decision makers, offering data that can be used to benchmark the results of a more focussed options analysis at the component level. Without some sort of broader benchmark, decision makers will invariably be challenged by the complex tradeoffs that are frequently identified in LCA studies spanning a broad range of environmental issues. For example, a 10% reduction in effluent nutrient levels might come at the cost of a 5% increase in energy use by the sewage treatment plant. But does that cause a 5% increase in the organisation's overall greenhouse gas (GHG) footprint, or a 0.5% increase? Might there be bigger opportunities for GHG mitigation in other components of the urban water system? Furthermore, are GHG emissions the only externality worth considering, or are there many more potential environmental pressures caused by this or other aspects of the whole system? Being able to answer such questions would help to prioritise across the environmental tradeoffs and management options under consideration.

Why another LCA study at the urban water system level ?

While a small number of previous studies have already explored life-cycle environmental burdens across full water and wastewater systems at the city scale (Amores et al. 2013, Barjoveanu et al. 2014, Friedrich et al. 2009, Lassaux et al. 2007, Lemos et al. 2013, Loubet et al. 2014, Lundie et al. 2004, Lundin and Morrison 2002, Mahgoub et al. 2010, Muñoz et al. 2010, Slagstad and Brattebø 2014, Uche et al. 2013), they will lack currency for many urban centres because of gaps in their scope. Most previous analysis has excluded one or more of the contemporary water supply technologies (e.g. rainwater tanks, seawater desalination, wastewater recycling) and wastewater treatment technologies (e.g. advanced biological nutrient removal) that are increasingly being adopted around the world. Nor have those studies fully utilised the wealth of recent knowledge generated on operational aspects that were, historically, not well recognised. Nitrous oxide emissions from sewage treatment (Law et al. 2012b), micropollutants in wastewater and biosolids, and energy use for small scale infrastructure (Vieira et al. 2014) are just some such examples.

Furthermore, a renewed research effort into life cycle impact assessment models has led to a number of improvements in model fidelity being implemented or recommended in recent years (see overviews in Finnveden et al. 2009, Rack et al. 2013, Zamagni et al. 2012). Of potential relevance to water

industry analysis are the efforts on impact assessment for water use (Kounina et al. 2013), eutrophication (e.g. Azevedo et al. 2013, Gallego et al. 2010, Helmes et al. 2012, Struijs et al. 2011), eco-toxicity and human toxicity (Hauschild et al. 2011), and resource depletion assessment (Klinglmair et al. 2014).

Only a subset of these developments have been considered in the most recent LCA studies of integrated urban water systems (e.g. Amores et al. 2013, Lemos et al. 2013, Muñoz et al. 2010), with further analysis required to understand the challenges and opportunities arising from these advances in LCA methodology.

Objective 1: *Generate comprehensive LCA results for a city-scale integrated urban water system, identifying which components of the greater system make the largest contributions to the overall results.*

Objective 2: *Identify the breadth of life-cycle impacts that could hold relevance for urban water systems planning.*

Should the life-cycle impacts be a concern for the urban water industry ?

Regardless of whether they are generated at the city-scale or treatment plant scale, LCA results do not of themselves indicate whether decision makers should care about the environmental issues that are assessed.

This is another substantial challenge when introducing LCA to a decision making environment, as the people involved will invariably know very little about some of the life-cycle environmental impacts (potentially) under consideration. For example, in the context of the urban water industry, planners will no doubt have a good understanding of local environmental concerns about freshwater extraction, and the eutrophication risks caused by nutrient discharge. In some parts of the world, their knowledge might also extend to the local toxicity risks associated with micropollutants. But they are less likely to have much understanding on whether toxicity impacts occurring somewhere else, at some point along the supply chain, should be considered important or not. In the case of other conventional LCA impact categories, they may have very little understanding at all.

In the LCA research community, one of the more conventional ways to gain this perspective is to normalise LCA impact results against some macro-scale benchmark. While this is relatively simple to do, given there are many such benchmark datasets available (Bare et al. 2006, Foley and Lant 2009, Laurent et al. 2011, Lautier et al. 2010, Sleeswijk et al. 2008), it carries some mathematical risk for the unsuspecting user. For any given benchmark, large differences in quality of the reference values can substantially bias the interpretation of normalised case study results (Heijungs et al. 2007). Since none of the published LCA benchmark datasets contain uncertainty estimates, it is very difficult for decision makers (or LCA practitioners) to understand where and how big these biases might be.

Furthermore, the nature and severity of bias will depend not just on the data gaps in the benchmark inventory, but also on how these align with data gaps or uncertainties in the case study inventory (Heijungs et al. 2007).

Even if those numerical risks can be overcome, the perspective provided by this conventional normalisation approach only goes so far, as the focus remains on environmental management largely for its own sake. That focus can provide only an indirect (if at all) indication on whether there is imperative for the urban water industry to be concerned about a particular issue.

Ultimately, the application of LCA will have the most influence if it identifies business risks that would have otherwise gone unrecognised. This is not a concept that has featured heavily in past scientific application of LCA to the urban water industry, however will be at front of mind for industries wishing to integrate life cycle principles into their decision making (e.g. Baitz et al. 2013).

The second part of this thesis uses this notion of 'business risk' to explore whether LCA's extended scope of environmental impact assessment might have some relevance for the urban water industry.

The use of LCA to identify business risks

As described above, the fundamental concern of environmental LCA is to identify 'environmental impacts' that might otherwise be overlooked in the decision making process. One way in which LCA addresses this goal is to introduce simplified metrics spanning a broad spectrum of environmental issues, allowing businesses to extend the scope of their environmental considerations beyond the bare minimum required to satisfy institutional (e.g. regulatory) constraints.

The most obvious point of such an exercise would be to identify any environmental problems that might become a direct concern for the industry, should there be a future shift in the policy landscape. Prudent decision making would avoid such risks if possible, particularly when the outcomes from that decision involve large commitments (e.g. capital) that deliver long term change and/or have irreversible consequences.

There are many past LCA studies highlighting such potential concerns for the urban water industry. For that reason, a focus of this thesis was to identify instances where conventional LCA practice might fail to adequately identify such risks.

Ozone depletion, LCA, and the urban water industry

The issue of stratospheric ozone depletion is one good example where conventional LCA methodology will fail to illustrate whether or not this should be considered important to the urban water industry.

The 1987 Montreal Protocol delivered widespread reduction in anthropogenic halocarbon emissions, and has been credited with preventing the complete collapse of the stratospheric ozone layer (Garcia et al. 2012, Newman et al. 2009). The remarkable success of this international response has encouraged the widespread perception that the ozone layer 'problem' has been 'fixed', and is no longer a prime concern for decision makers not directly involved in halocarbon emissions management.

In contrast, the atmospheric science community remains very focussed on the new challenges associated with longer term ozone layer recovery and management. The dominant drivers of ozone layer status through the 21st century will in fact be the growing emissions of more conventional greenhouse gases. Anthropogenic N_2O emissions are now seen as the biggest threat to ozone layer recovery, already more substantial than the remaining halocarbon emission sources (Daniel et al. 2010, Ravishankara et al. 2009). On the other hand, CH_4 and CO_2 have a positive influence on stratospheric ozone levels that is expected to more than counteract the influence of N_2O (Fleming et al. 2011, Plummer et al. 2010, Revell et al. 2012), Even conservative forecasts of GHG emissions growth predict an ozone layer 'super recovery' - whereby global ozone abundance greatly exceeds the levels found in pre-industrial times (Austin et al. 2010, Eyring et al. 2010, Fleming et al. 2011, Oman et al. 2010, Plummer et al. 2010).

Ozone depletion assessment has a long history in LCA, with midpoint analysis favouring the use of steady-state *Ozone Depletion Potential* factors to reflect relative changes in ozone abundance.

Unfortunately, the LCA research community has been very slow to pick up on these newer developments in ozone layer science, overlooking them completely in recent recommendations on 'best practice' for impact assessment in LCA studies (Hauschild et al. 2013). This raises the possibility that continued use of the conventional LCA models could completely overlook those emissions that might otherwise be considered most important by the atmospheric science community. This could lead to a somewhat perverse outcome, whereby the use of LCA would reinforce, rather than challenge, the common but mistaken view that the ozone layer is returning to 'normal' now that halocarbon emissions are under control.

It remains to be seen how the international policy community will respond to the increased scientific focus on the role these greenhouse gases (N_2O, CH_4, CO_2) have in dictating the future of the ozone layer. As the 2010 Scientific Assessment of Ozone Depletion by the World Meteorological Association (WMO 2011) was the first to include substantive coverage of the implications for ozone layer protection from non-halocarbon emissions, it may be that it will take until the next 4-yearly assessment before a more focussed set of recommendations are developed. In the meantime,

however, debate on the merits of including N_2O into ozone layer protection strategies has already begun (Kanter et al. 2013).

Since all three of these gases feature prominently in debates about the GHG implications of the urban water industry, their inclusion in ozone layer assessment metrics could provide useful insight into the implications of any changes in the ozone layer management policy landscape.

> ***Objective 3:***
> *Identify whether or not there is a case for the LCA community to modify its approach to assessing impacts on the stratospheric ozone layer.*
>
> *Explore whether an international policy shift in ozone layer management could have practical implications for the urban water industry.*

Business risks associated with the use of LCA

Whether or not the increasing use of LCA represents an advance or a distraction for human endeavour, that debate becomes somewhat of a moot point if its influence in everyday decision making continues to grow. Such an outcome introduces another potential source of business risk, should the misuse of life cycle impact assessment (LCIA) unreasonably cast a negative light on some particular product or service.

Biosolids, LCA, and the agricultural industry

Biosolids management is one element of the urban water industry where such a risk is particularly worth considering. The disposal of waste biosolids is a perennial, and growing, challenge, with water utilities in many parts of the world being heavily reliant on agricultural producers to absorb large quantities of waste biosolids.

In Australia, nearly 60% of the 330kt/yr of waste biosolids produced from sewage treatment plants (STPs) are applied directly onto agricultural fields (ANZBP 2013). Furthermore, the Australian water industry is actively promoting further growth in this approach, so as to mitigate growing risks for utilities reliant on landfilling or stockpiling of such large volumes of STP waste product.

This policy position is underpinned by a decade-long local research endeavour, which has concluded that appropriately managed biosolids use can deliver benefits at the field (soil condition; crop yield), at low risk to human health and the local environment (Pritchard et al. 2010). Research and policy attention is also now turning to the question of whether biosolids use on farms can deliver substantial longer term benefits by sequestering carbon in the soil, or alleviating potential supply shortfalls in the global phosphorus cycle (AWA 2012).

Other parallel developments might also have some bearing on the enthusiasm for biosolids use on Australian farms. With so many of Australia's agricultural sectors being dependent on exports to foreign countries, the industry is embracing the use of Life Cycle Assessment (LCA) as an analytical tool. Internationally, there is a strong trend towards the use of LCA to support decisions made on the choice of food imports. Driving this is a mix of consumption-based national GHG initiatives (Barrett et al. 2013), more general policy imperatives (Wolf et al. 2012), high profile (albeit sometimes transient) consumer concerns (e.g. food miles), and the more substantial shift towards incorporating metrics of 'environmental footprint' on food product labelling.

The majority of LCA applied to biosolids has in fact been from the perspective of the water industry, used specifically to compare different biosolids disposal options (see Yoshida et al. 2013). In that body of work, there has been very little critique applied to how well the issues relevant to agricultural use of biosolids can actually be characterised. Instead, the majority of analytical focus is directed towards engineered treatment and disposal technologies for managing the solids stream (Yoshida et al. 2013).

Taken at face value, this suggests the possibility of a growing risk for urban water utilities. In Australia at least, farmers have until now been generally quite willing to utilise STP biosolids where they are available – they are typically provided at no cost[1], and save them expenditure on conventional fertilisers. However, if LCA gains traction as a decision support tool for use in the agricultural sector, then closer attention might be paid to whether or not biosolids use compares favourably on environmental grounds, when compared with the use of alternative fertiliser sources.

> ***Objective 4:***
> *Investigate whether the use of contemporary LCA impact assessment models in the Australian agricultural sector would support the continued practice of applying waste biosolids to agricultural fields.*

[1] In some regions of Australia, water utilities will pay farmers to take their biosolids. In other regions, the biosolids are provided free of charge.

3. THESIS OVERVIEW AND RESEARCH METHODS

Overview

The four objectives are delivered through an interconnected set of literature reviews and case studies (Table 1).

Table 1: Overview of thesis structure

Obj-ectives	Case Study	Research Step	Thesis sections	publication
1 & 2	#1 Analysis of urban water system scenarios	overview, results & conclusions	Appendix B1	*Lane, J.L., de Haas, D.W., Lant, P. (under review) The diverse environmental burden of city-scale urban water systems; Submitted to Water Research*
		scenario design & data collection	Appendix B2	
		impact assessment model definition		
3	--	background literature review	Appendix A1	*Lane, J., Lant, P. (2012) Including N2O in ozone depletion models for LCA. The International Journal of Life Cycle Assessment 17, 252-257.*
		significance for water industry LCA	Appendix A2	*Lane, J. (in press) Stratospheric Ozone Depletion. Chapter 5 in 'Compendium of Life Cycle Assessment'. (Eds W Kloepffer and M Curran) Vol.4, Life Cycle Impact Assessment - LCIA. Springer.*
4	#2 Analysis of biosolids application to farmland	scenario design & data collection	Appendix C1	*Lane, J.L., Lant, P. (in prep) Biosolids and agriculture – a growing 'LCA risk' for urban water utilities.*
		develop characterisation factor for phosphorous resource depletion		
		other impact model definition		
		detailed GHG and energy analysis		
		assessment of policy risk		

3.1 Case study development

Two case studies are presented, both derived from infrastructure systems in place at the Gold Coast region of Australia. The first provides macro-level analysis of the overall urban water system for the Gold Coast city, presented in Appendix B. The second (Appendix C) provides more focussed analysis of biosolids disposal challenges, improving on key assumptions, and extending the scope to include additional issues of particular relevance to analysis undertaken from the perspective of the agricultural sector.

In both cases, the focus of the analysis is to develop generalised conclusions of relevance to urban water managers across the country, rather than to comment explicitly on decision making options specific to the Gold Coast. The Gold Coast water system infrastructure comprises many of the elements (advanced wastewater nutrient removal; large scale biosolids reuse; a diversified water supply system) of interest to urban water system planners in other regions. Using detailed data for this infrastructure provided a robust underpinning for the policy level considerations focussed on in this thesis.

Case Study #1 - Urban water system scenarios

Two urban water system scenarios were developed, collectively encapsulating the technologies in use (or under consideration) in urban centres across much of the developed world (Figure 1).

The **'Traditional infrastructure' scenario** represents the 'linear' model that is typical of urban centres in most developed countries. This comprises dam-sourced mains supply (83% of total supply) supplemented with a small number of household rainwater tanks (2%); centralised sewage collection and treatment with advanced biological nutrient removal (BNR) processes; discharge of treated wastewater to local waterways with a portion reused for non-residential irrigation (15% of total water supply); and biosolids sent to agricultural reuse.

Water and sewage balances for the 'Traditional infrastructure' scenario were determined by calibrating top-down and bottom-up approaches to derive assumptions for average water supply and demand. The top-down approach utilised city-scale water supply information (NWC 2009). The bottom-up approach was based on local empirical data for household water end-use (Willis et al. 2009, Willis et al. 2010) combined with rainwater tank yield modelling undertaken for this study.

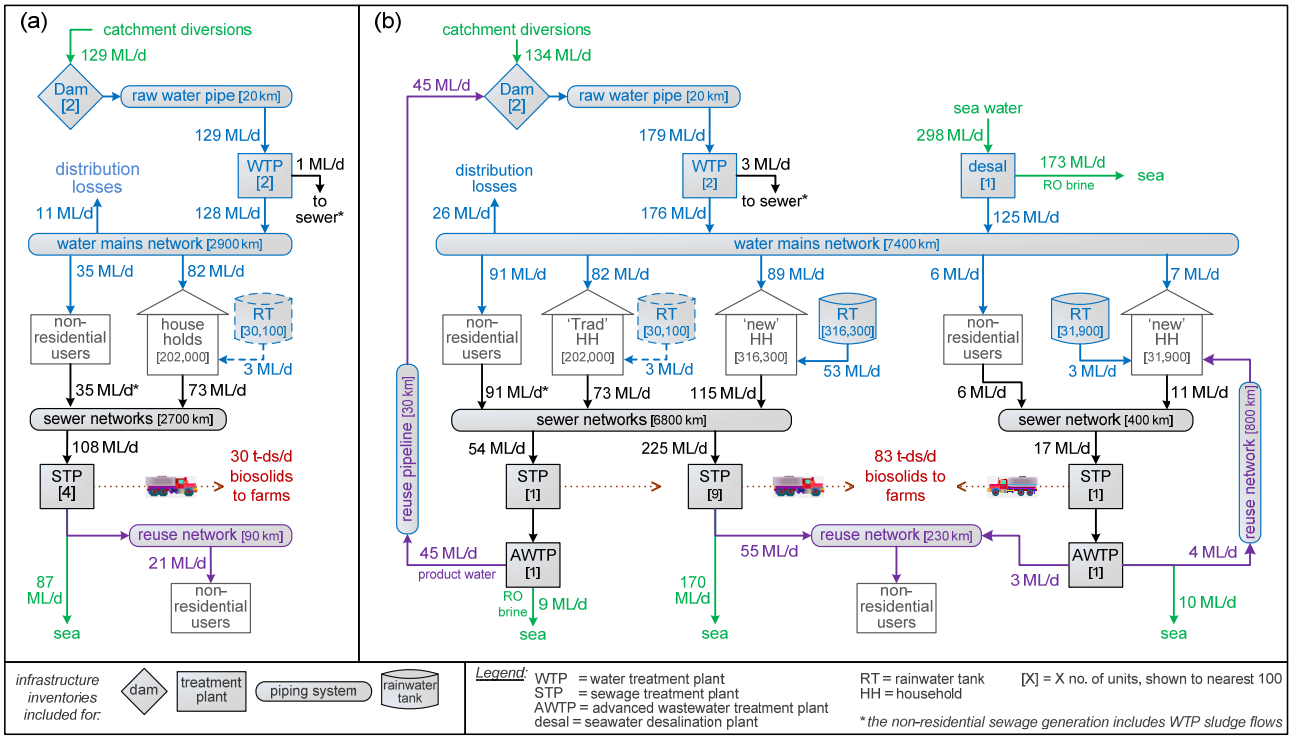

Figure 1: flowsheets for the (a) 'Traditional infrastructure' scenario, and (b) 'Diversified infrastructure' scenario, showing the main process flows and the number of units [X] for each system component. The difference between total water use, and total water entering the sewer system, is the water used for irrigation and other external purposes.

The **'Diversified infrastructure'** scenario incorporates a range of different urban water supply technologies: increased supply from household rainwater tanks (15% of total supply); sea water desalination (29% of total); indirect potable wastewater recycling (10% of total); non-potable direct

wastewater reuse to the residential sector (2% of total). Non-residential, direct reuse of treated wastewater was scaled up to remain at 15% of the supply mix. While this scenario also incorporated a small increase in yield from the existing Gold Coast dams, the contribution from dam-based supplies is now only 30% of the total supply mix.

So as to focus on the implications of this change in water supply technologies, all other changes to the system configuration of the 'Diversified infrastructure' scenario were kept to a minimum. Since a substantial increase in overall water supply capacity was introduced, the population base was also increased. This allowed consistency in the rate of per-capita water end-use across the two scenarios. Also kept constant were the per-capita sewage characteristics (flow, COD, nutrient loads), and the overall wastewater management strategy (technology type; effluent and biosolids concentrations; effluent and biosolids discharge points) - other than for those changes induced by the adoption of wastewater recycling.

Case Study #2 - Biosolids disposal scenarios

The second case study reviews the implications of changing the method of biosolids disposal, from some default approach, to the application of biosolids to farmland (Figure 2). This change in disposal options is not just relevant for decision making in the urban water industry, but also for the agricultural sector. Biosolids generation rates will typically be dictated by STP effluent quality requirements, and the influence this has on the choice of STP treatment technologies. Because of this, the decision by a farmer to begin (or cease) using biosolids as a soil supplement will influence the disposal pathway chosen for that biosolids, rather than the amount of biosolids produced.

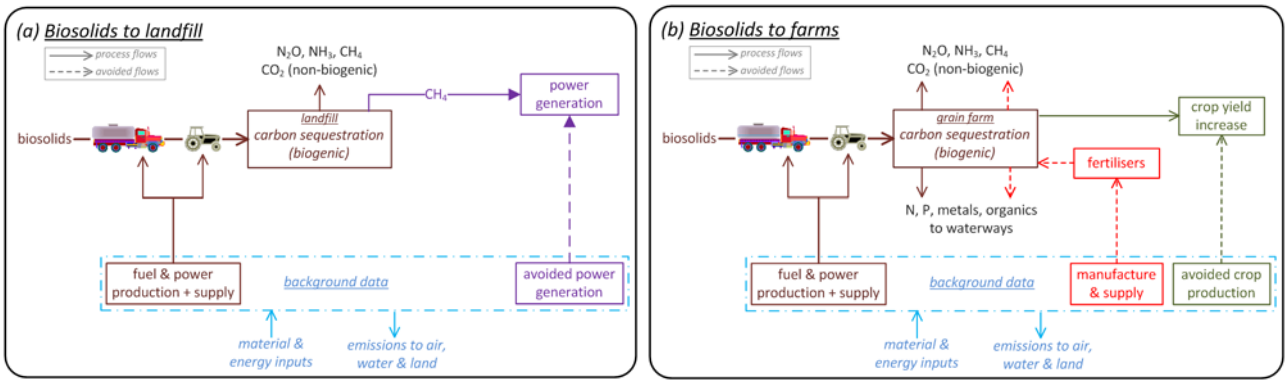

Figure 2: System boundary for the scenarios involving (a) disposal in an urban landfill site; and (b) direct application of biosolids to agricultural land. Solid lines represent flows that are attributed to the biosolids disposal. Dotted lines represent flows that are avoided by the use of biosolids, and allocated as a credit to the scenario results.

For this study, the default disposal method is assumed to be landfilling. The decision on whether to shift from such a practice, to one where the biosolids is used as a direct supplement on agricultural fields, is a choice being made or considered by many urban water utilities around Australia.

The modelling for this exercise is also undertaken at the city scale – i.e. the biosolids quantites are the same as in Case Study #1. This allows the results to be benchmarked against the overall life-cycle environmental burden associated with the urban water system developed in case study 1.

This benchmarking step provides useful perspective on the potential significance of these results to water utilities. To provide the equivalent perspective for agricultural producers, the results are also benchmarked against an estimate of the life-cycle impact results associated with crop production on the biosolids-amended fields.

So as to focus on the agricultural-use aspect of the biosolids system, two important simplifications are made in defining the system boundary of this study:

- Firstly, any (potential) need for additional STP processing to stabilise or dewater the biosolids is excluded, essentially assuming that the STP waste product in the baseline (landfill) scenario already meets the local quality standards for application to agricultural fields. Depending on the technology employed, such treatment could be a substantial component of the LCA 'burden' for the overall biosolids system (Peters and Rowley 2009). For future LCA case study analysis serving an actual decision purpose, it would not be valid to exclude the treatment implications without careful scrutiny. With sufficient engineering input, modelling these treatment options can be undertaken with a relatively high degree of confidence. However, such treatment implications are excluded in the analysis provided for this thesis, to allow a greater focus on aspects of biosolids management that are less well understood and harder to quantify (e.g. the response of agricultural soils).
- Secondly, the analysis is also limited to changes in system operations, on the premise that minimal additional on-farm physical infrastructure would be required to facilitate the agricultural use. The possibility of changes in the physical infrastructure or land footprint required for urban landfill is also not considered. These exclusions are unlikely to affect our conclusions, with previous studies identifying the minor impacts associated with the infrastructure construction life-cycle stage (Peters and Rowley 2009).

The system boundary does incorporate the implications of offsets generated by the disposal of the biosolids:

- For the landfill option, this accounts for the methane produced from degradation of the biosolids carbon, then used to generate electricity.
- For the agricultural use option, the reduced need for manufacture and supply of synthetic fertilisers is credited to the final results.

A credit was also given for crop yield increases attributed to the biosolids application. Certain Australian studies have shown biosolids amended fields to achieve greater crop yields than comparable fields fertilised with conventional, synthetic products (Barry and Bell 2006, Powell and Graham 2012). However, such effects vary substantially in the small amount of data available, with other studies showing no statistically significant benefit at all (e.g. Ives et al. 2011, Nash et al. 2011). This variability was considered in the sensitivity analysis undertaken throughout the biosolids analysis.

3.2 Generation of inventory data

Overview

Operational data for each of the system component technologies was compiled from a range of sources, combining locally measured data with the best available sources of empirical and literature based information.

Infrastructure construction inventories were included in Case Study 1, but excluded from Case Study 2.Infrastructure construction data was in most cases taken from a variety of non-local sources, and annualised based on estimates of equipment lifespan. The exception was for the water and sewerage networks, where detailed piping inventories provided by the local Gold Coast water utilities were combined with assumptions on pipeline construction developed for a previous Australian study (Sharma et al. 2009).

Data for second order inventories (e.g. power generation; materials supply) were taken, where possible, from the AusLCI (2012) and Life Cycle Strategies (Grant 2012) Australian LCI databases. Otherwise they were based on Ecoinvent data (Frischknecht et al. 2007). Electricity supply was modelled as the average of the generation mix in the state of Queensland, which (for the inventories used in this study) is dominated by black coal (85% of total) and natural gas (12% of total) combustion plants.

Modelling the infrastructure operations

Treatment plant models for the 'Traditional infrastructure' scenario were based on analysis of detailed monitoring data collected for the relevant treatment plants at the Gold Coast, for at least a 12 month period up to and including 2008. Where relevant, equivalent estimates for the 'Diversified infrastructure' scenario were obtained by scaling up the model inventories in proportion to flow and/or population increases. System models for desalination and direct non-potable wastewater reuse were based on more recent monitoring data from Gold Coast systems that have been commissioned since 2009. Modelling of the indirect potable (wastewater) reuse system was based on detailed

monitoring data from an equivalent system in a nearby region, adjusted for changes in influent nutrient concentrations, and using distribution system designs (for product water and brine disposal) relevant to this study region.

Estimates for other key flows were informed by the latest science in each of the relevant areas, supplemented with expert opinion as required.

Electricity use for household rainwater tank delivery pumps was calculated by compiling empirical, end-use specific, data from a range of recent Australian studies (Beal and Stewart 2011, Hauber-Davidson and Shortt 2011, Siems et al. 2013, Tjandraatmadja et al. 2013).

A detailed literature review was undertaken for direct emissions of N_2O, CH_4 and non-biogenic CO_2 from the water system infrastructure. Empirical evidence specific to the Gold Coast area was used to estimate CH_4 emissions from sewer systems (Foley et al. 2009, Guisasola et al. 2008, Liu et al. in prep) and from water supply reservoirs (Grinham et al. 2011, Sherman et al. 2012). Data from a neighbouring region, with similar urban characteristics, was used to quantify the influence of non-biogenic sewage carbon (Law et al. 2013). More general literature-based data was used for other gas flux estimates.

Locally relevant industry data was used to estimate micropollutant concentration profiles for metals and organics in biosolids, metals in wastewater, and organics in recycled wastewater streams. For organic micropollutants in secondary treated wastewater, a hypothetical profile was developed using literature data from the local region (Reungoat et al. 2010, Watkinson et al. 2009).

Modelling the biosolids disposal

In all cases, biosolids composition (moisture, N, P, C, trace metals and organics) is set to equal the average (weighted by total mass of dry solids) of the available data for the four STPs in the *'Traditional'* scenario of Case Study #1. The contaminants (metals and organics) profile is satisfactory to comply with the requirements for agricultural biosolids reuse in the local jurisdiction (EPA 2002).

For **Case Study #1**, the assumptions used here are in line with the majority of previous, generalised LCA analyses of biosolids application to crops. Fertiliser displacement ratios were calculated using generic bioavailability factors from Foley et al. (2010a).

Biosolids nutrient and carbon content based on mass balances, with fertiliser displacement calculated using bioavailability factors from Foley et al. (2010a).

Case Study #2 involved a number of modifications to the operational inventories, with sensitivity analysis used to explore the implications of uncertainty or variability in certain key assumptions. For

this, the system boundary was extended to also include a crop yield response due to the biosolids application, based on data from the agricultural zone that accepts biosolids from the Gold Coast treatment plants (Barry and Bell 2006). To minimise any methodological inconsistencies, a more deterministic set of nutrient balance relationships was used that align with the data taken from those same field trials.

Case Study #2 also required operational inventories developed for the option of biosolids disposal to landfill. Modelling of the landfill processing systems (power and fuel use; power generation) was based on the generic landfill inventories contained in the Australasian life-cycle inventory database of Life Cycle Strategies (Grant 2012). The landfill carbon balance was calculated with the 1st order decay model recommended for use in GHG accounting for Australian landfill sites (DCCEE 2011), assuming a notable portion of the biosolids biosolids carbon is of fossil origin (see Section 4.1). It was assumed the landfill would be fully sealed, preventing any leaching of contaminants to groundwater systems.

3.3 Selection of impact assessment models

This analysis follows the midpoint approach to Life Cycle Impact Assessment (LCIA). Midpoint indicators are measures of *potential* impact, defined at some intermediate point along the cause-effect chain from intervention (e.g. emission) to actual impact of concern (e.g. biodiversity; human health). While these midpoint-level indicators are necessarily incomplete in their description of environmental pressures, they often have the advantage of aligning well with metrics used for informing policy making.

The choice of impact assessment models was tailored for relevance to urban water systems, in accordance with the principles specified in ISO14040 (2006a).

Case Study #1 used a deliberately broad set of 14 impact categories, spanning environmental, resource depletion and human health issues. The impact assessment models were taken from a range of sources, giving consideration to the recommendations for European practice from a review of the best available models in 2009 (Hauschild et al. 2013), and a review of more recent literature for the priority areas in this study (Section 4; Appendix A). For the *Ozone Depletion* impact category, a detailed literature review was undertaken to justify the choice of a non-conventional characterisation factor for N_2O emissions. Models from the ReCiPe suite (Goedkoop et al. 2009) were adopted by default in four other cases, as this provides one of the most comprehensive, and extensively published, set of LCA impact categories available.

Only a subset of these were used in **Case Study #2**, focussing on seven impact categories that (a) were indicated in the results of Case Study #1 to be of relevance to biosolids management analysis; and (b) have some overlap with the scope of Australian research and/or debate on biosolids use for agriculture. Furthermore, three of these align with the nominated priority categories for agricultural LCA development in Australia (Eady et al. 2012, Eady et al. 2014), while the other four are some of the most commonly used in the LCA domain.

For Case Study #2, modifications were made to certain impact assessment models so as to provide more focussed analysis of the biosolids disposal issue. A more generic metric was used for assessing the potential aquatic eutrophication impacts, so as to avoid the influence of choosing locationspecific nutrient sensitivities. For assessing the significance of longterm minerals resource depletion, an alternative metric was compared and contrasted with the approach used in Case Study #1. The base metric was taken from the ReCiPe suite (Goedkoop et al. 2009), with the addition of a characterisation factor for mineral phosphate resources derived in this thesis.

Table 2: Description of impact categories considered in this thesis

Indicator		as proxy for…	Case Study
FES	Freshwater Extraction Stress	Ecosystem impacts from disruptions to the hydrological cycle	1
FEU	Freshwater Eutrophication	Ecosystem impacts from nutrient enrichment and oxygen depletion in waterways	1,2
MEU	Marine Eutrophication		
MTX	Marine Ecotoxicity	Ecosystem and human health impacts of chemical emissions to air, water and land	1
FTX	Freshwater Ecotoxicity		1
TTX	Terrestrial Ecotoxicity		1,2
HTX	Human Toxicity		1,2
TA	Terrestrial Acidification	Ecosystem impacts from soil acidification	1
POF	Photochemical Ozone Formation	Human health impacts from lower atmosphere ozone buildup (POF), & inhalation of particulate matter (PF)	1
PF	Particulates Formation		1
GW	Global Warming	Human & ecosystem impacts from climate change (GW), & depletion of the stratospheric ozone layer (OD)	1,2
OD	Stratospheric Ozone Depletion		1,2
FFD	Fossil Fuels Depletion	Resource availability for future generations	1,2
MD	Minerals Depletion	Resource availability for future generations	1,2
CED	Cumulative Energy Demand	Total energy footprint	2

4. LITERATURE REVIEW

Overview

This thesis involved extensive use of literature review, both to scope and design the studies, and also to provide data and models as inputs to the analysis.

Chapter 2 describes the literature base underpinning the formation of objectives for this thesis. More detailed background reviews are provided in the introduction to the manuscripts provided in 0, Appendix B1 and 0. Those messages are not repeated here.

Other more technical literature reviews were undertaken to define certain inputs to the modelling methodology. These reviews underpinned a number of key assumptions used in generating the system inventories for the two case studies, and the choice of impact assessment models. These modelling inputs are described in full in Appendix B2 and 0.

This section provides a brief summary of the reviews undertaken to develop some of those inputs that feature more prominently in the conclusions drawn from this thesis:

(i) estimating direct emission of greenhouse gases (CO_2, CH_4, N_2O) from urban water system operations;
(ii) assessing the significance of GHG emissions in terms of their effect on the stratospheric ozone layer; and
(iii) assessing the significance of phosphorus recovery in terms of minerals resource sustainability.

4.1 Estimating direct greenhouse gas emissions

This section draws on the literature review in the manuscripts:

(Appendix B): Lane, J.L., de Haas, D.W., Lant, P. (under review) The diverse environmental burden of city-scale urban water systems. Submitted to Water Research.

(Appendix C): Lane, J., Lant, P. (in prep) Biosolids and agriculture – a growing 'LCA risk' for urban water utilities.

Direct emission of CH_4 – water supply dams

While dam methane (CH_4) generation has been extensively studied in a small number of water supplies around the world (Barros et al. 2011, Bastviken et al. 2011), there is very little information available to guide a more generalised assessment on the significance of CH_4 fluxes from specific urban water supply dams.

The Gold Coast is somewhat of an anomaly in this regard, where two recent studies have collected data from the Hinze (Sherman et al. 2012) and Little Nerang (Grinham et al. 2011, Sherman et al.

2012) dams. The available evidence indicates that these emissions are predominantly caused by ongoing carbon inputs (e.g. leaf litter) to the water bodies, rather than from the vegetation originally inundated at the time the dam was first constructed.

A methane emission estimate for the Little Nerang Dam (268 mg-$CH_4/m^2/d$) was set to equal the average of the three daily average flux rates from Grinham *et al.* (2011), and a separate estimate from Sherman et al. (2012). The equivalent emission factor for Hinze Dam (44 mg-$CH_4/m^2/d$) is taken from Sherman et al. (2012). Sherman et al. (2012) also reported a range of other area-weighted average values for both dams, using different methodological approaches to deal with sampling bias. Only the higher end estimates were used here, on the premise their sampling approach may have underestimated ebullition (bubble fluxes) from these storages, thereby giving undue weighting to the much lower diffusive flux rates that also occur. The Grinham *et al.* (2011) study, and other local data (Grinham 2012), indicate that intermittent bubbling rates can be substantial in water bodies with forested catchments such as those at the Gold Coast. Of the two dams, the Little Nerang catchment has a much higher degree of vegetation cover, presumed to be the reason for its much higher specific CH_4 flux rates.

To calculate the overall (average annual) CH_4 flux, those emission factors were combined with estimates for water body surface area at the nominal Full Supply Level. For Little Nerang Dam, a surface area of 49 ha was used in both scenarios. For the 'Traditional Infrastructure' scenario, the surface area of Hinze Dam was set at 972 ha, as per the specifications prior to the dam wall raising.

For the 'Diversified infrastructure' scenario, which incorporates the dam raising, the CH_4 flux estimate was increased by 1.24 times, being the square root of the 55% increase in Full Supply Level surface area after the dam wall was raised. This assumes that overall bubble-sourced emissions might increase in proportion to increases in the water body circumference, given the observation that bubble flux emissions occur predominantly at shallow depths at the periphery of the water body (Grinham et al. 2011).

Despite the availability of local empirical data, the estimates for dam-generated CH_4 should be considered as highly uncertain. There is scant information available to guide the prediction of CH_4 flux rates in response to changes in management strategy for an existing dam (such as the dam wall raising considered here). Furthermore, the available monitoring studies report gross emissions only, and there remain gaps in understanding what level of net anthropogenic emissions could be attributed to the existence of the water supply dams.

Biogenic versus non-biogenic carbon in sewage

The conventional approach for water industry analysis is to assume that all sewage carbon is biogenic in origin, meaning that downstream mineralisation of this carbon is ignored in GHG accounting protocols (IPCC 2006).

However four recent studies have identified notable levels of non-biogenic (e.g. fossil) carbon in urban wastewater systems (Griffith et al. 2009, Law et al. 2013, Nara et al. 2010, Yoshida et al. 2014). The origin of this non-biogenic is thought to be surfactants and other chemical products (Griffith et al. 2009, Law et al. 2013), with substantial contributions attributed to industrial wastewater discharges into the sewer (Law et al. 2013). Mineralisation of any such carbon in the wastewater treatment process should be included in life-cycle based GHG footprints.

For this study, 10% of all sewage carbon is assumed to be of non-biogenic origin. That estimate is the load-weighted average of measured data for four sewer systems in regions neighbouring the Gold Coast city (Law et al. 2013). For each STP in our scenarios, an overall ratio of biogenic : non-biogenic was calculated across all the carbon inputs, then applied uniformly to all carbon outflows (as CO_2; as CH_4; in the wastewater; in the biosolids) from that plant. This also accounted for the origin (biogenic vs. non-biogenic) of any chemically dosed carbon sources into the STP.

To our knowledge, the locally relevant data from Law et al. (2013) are the most comprehensive such values reported in the literature. Their range (4-14% of carbon is non-biogenic) agrees well with the single data point (14% of carbon in STP influent is non-biogenic) collected by Yoshida et al. (2014). Further field studies would be required to ascertain whether the assumptions used here are representative of other regions in the world.

Much higher fractions of non-biogenic carbon (25% of dissolved organic carbon; 14% of particulate organic carbon) were detected in another study focussed on treated effluent from activated sludge plants (Griffith et al. 2009), and occasionally used for the sensitivity analysis in urban water system studies (Carballa et al. 2011, de Haas et al. 2014, Rodriguez-Garcia et al. 2011). However, the utility of that data for quantifying overall non-biogenic carbon loads is questionable. Typically, residual carbon levels in such effluent streams are very low – in our study, the carbon in treated wastewater constitutes only ~5% of the total sewage carbon load. Sewage based data points (as used here) are therefore more likely to be useful for understanding the source of carbon in the wastewater system.

Direct emission of CH_4 – sewer systems

Substantial CH_4 generation has been observed in sewer networks by a number of recent Gold Coast studies (Foley et al. 2009, Guisasola et al. 2008, Guisasola et al. 2009), and in the raw sewage entering

STPs in other Australian and international locations (Daelman et al. 2012, Law et al. 2012a, Wang et al. 2011).

Sewer CH_4 generation rates would be expected to vary substantially across different systems and locations. The main cause of sewer generated CH_4 is thought to be anaerobic bacterial activity occurring in rising mains, with the generation rate being sensitive to (amongst other things) residence time and temperature (Foley et al. 2009, Guisasola et al. 2009, Liu et al. in prep). The presence of trade waste inputs with highly biodegradable carbon could increase the level of CH_4 production (Sudarjanto et al. 2011), whereas certain chemicals commonly dosed to control corrosion have been shown to inhibit CH_4 production (Jiang et al. 2013, Zhang et al. 2009). While there is also some evidence of CH_4 generation in open gravity systems (e.g. Foley et al. 2009), this phenomenon is less well understood and characterised in the literature.

Given the prevalence of pressurised (rising main) sewers in the Gold Coast sewer network, sewer generation of CH_4 in both scenarios was set to equal 5 mg-CH_4 per litre of total dry weather flow. While a somewhat arbitrary choice, this 'emission factor' represents a conservative mid-range estimate when applied across the entire sewer system for this case study. 5 mg-CH_4/L is at the lower end of measured values from the distinct sampling campaigns conducted on Gold Coast rising mains (Foley et al. 2009, Guisasola et al. 2008, Liu et al. in prep), with dissolved CH_4 concentrations as high as 25 mg-CH_4/L being detected. It is also substantially lower than the prediction of ~12mg/L by local modelling work (Foley et al. 2009, Guisasola et al. 2009), considered to be a realistic estimate for large scale pressurised sewer systems in Australia (Foley et al. 2009).

The final component of this calculation was to assume that 100% of the sewer generated CH_4 is stripped to atmosphere. No data or models were found in the literature that could support a more precise assumption. In one study, the activated sludge process oxidised up to 80% of the dissolved CH_4 in the plant feedstock, thereby avoiding a (potentially) substantial emission burden (Daelman et al. 2012). In contrast, results for one particular Australian STP indicate a much higher fraction of dissolved CH_4 in the raw sewage being stripped during the STP operations (Law et al. 2012a). Furthermore, both those studies are limited to analysis of the dissolved CH_4 reaching the STP, but provide no indication of how much sewer generated CH_4 might already have been stripped at release points (e.g. pump stations) throughout the sewer network.

Carbon sequestration - biosolids application to farm soils

The long-term sequestration of biogenic-sourced carbon would represent a net removal of CO_2 from the atmosphere, and therefore should be given a credit in life-cycle based GHG analysis. In contrast,

the sequestration of fossil (non-biogenic) carbon should be excluded, since this represents no net change to the concentration of radiative forcing agents in the atmosphere.

For this study, the sequestration potential is taken from the default approach (70 g-C / kg-dry solids applied to soils) provided for use by the Canadian water industry (Brown et al. 2010). This translates to approximately 24% of the biosolids carbon being retained in the soil over the long term. To quantify the overall implications for the GHG footprint, that carbon sequestration estimate is combined with the assumptions used on the biogenic vs. non-biogenic carbon split in biosolids (see above).

The results from this approach should be interpreted with caution, since there are multiple reasons why this sequestration factor might lack relevance for many parts of the world. Soil carbon stability is likely to be dependent on climate, agricultural management practices, and a host of other mitigating factors (Baldock et al. 2012, Luo et al. 2010, Sanderman and Baldock 2010, Thorburn et al. 2013). In the Australian context, for example, carbon sequestration from the direct addition of organic matter to local soils might well be lower than in equivalent overseas studies, given that the warmer Australian climates will encourage higher soil degradation rates (Sanderman and Baldock 2010, Sanderman et al. 2010). Furthermore, the quality of predictive soil carbon estimates will be constrained by challenges in interpreting the limited availability of longitudinal studies indicating the extent to which soil-sequestration can be maintained over the long term (Sanderman and Baldock 2010). Specifically in relation to biosolids application, the limited available field data is ambivalent on whether increases in soil carbon will be maintained beyond the short term (Barry and Bell 2006, Bolan et al. 2013, Ives et al. 2011, Powell and Graham 2012).

Given these concerns, the simplified estimate used here is likely to represent an upper bound of sequestration possible from biosolids application to Australian soils. It is not possible to determine a more realistic set of assumptions, hence any analysis focussed on biosolids carbon sequestration will involve a high degree of inherent uncertainty.

Direct emission of N_2O – sewage treatment plants

The potential for notable levels of N_2O emissions from urban wastewater treatment systems has long been recognised (Czepiel et al. 1995). Until recently, however, there has only been scant information available on which to base the quantitative estimates required for GHG footprinting and LCA studies.

Recent years have seen a significant increase in research effort both to quantify N_2O emissions from STPs, and to understand the fundamental mechanisms leading to N_2O creation. This has exposed the inadequacies of conventional N_2O accounting methodologies. For example, a number of studies have estimated N_2O emission levels an order of magnitude higher than would be calculated using the

default methodology proposed by the IPCC (see Daelman et al. 2013b, Kampschreur et al. 2009). In the Australian context, the mandated approach (DCCEE 2011) and recent literature (de Haas et al. 2009, Foley et al. 2010b) relate N_2O flux to the level of denitrification that is achieved, reflecting the historical perception that the denitrification pathway was the main source of N_2O from STPs. However, recent investigations have shown that process conditions affecting the nitrification pathway can be an important determinant of overall N_2O emission rates (Ahn et al. 2010, Law et al. 2012b, Ni et al. 2013, Sun et al. 2013, Ye et al. 2014). Denitrification based emission factors are therefore not likely to correlate consistently well with the key causative factors of N_2O generated in wastewater treatment processes.

There are now quite a number of published emission estimates for full scale STPs, covering a variety of process configurations and process conditions (see Desloover et al. 2012, Law et al. 2012b). While it is clear that the type of STP configuration could play an important role in determining the scale of N_2O emissions (Ahn et al. 2010, Foley et al. 2010b), there is not enough literature data available to meaningfully select emission estimates specific to the STPs included in this case study. Firstly, this is because the scale of emissions can also be strongly influenced by the variation in process operating conditions that will exist across similar installations (Ahn et al. 2010, Foley et al. 2010b, GWRC 2011, Law et al. 2012b). Secondly, recent studies have illustrated the extent to which N_2O fluxes can vary over diurnal to seasonal timescales (Aboobakar et al. 2013, Ahn et al. 2010, Daelman et al. 2013a, Sun et al. 2013, Ye et al. 2014) - casting doubts over the representativeness of most previous analysis based on short term sampling regimes. Others have identified another potential source of underreporting, claiming the physical sampling apparatus used in some previous studies was unlikely to have effectively captured N_2O from the major emission zones (Ye et al. 2014).

While the available data generated in recent years has clearly helped to highlight the potential scale of this emission source, it is somewhat unreliable for the purpose of proposing generic emission factors for large scale STPs (Aboobakar et al. 2013, Ahn et al. 2010, Daelman et al. 2013b). Given this, the estimate for this study was based on recent data collected across a large number of sites, rather than extracting information from any one particular example. Overall estimates were considered for 30 full scale, urban STPs, covering a range of process configurations and conditions (Aboobakar et al. 2013, Ahn et al. 2010, Daelman et al. 2013b, Foley et al. 2010b, GWRC 2011, Joss et al. 2009, Kampschreur et al. 2008, Ni et al. 2013, Sun et al. 2013, Ye et al. 2014). The average (15.5 g-N_2O/kg-TN in the influent) of those 30 flux estimates is double the median value (7.5 g-N_2O/kg-TN in the influent), illustrating the importance of recognising and understanding the variation in the published data. High uncertainty will be associated with any emission factors derived from the available data.

To allow for the possibility that some of the higher estimates in this set might reflect transient conditions that are not realistic over the longer term, the default emission factor was set at the median of the data surveyed (7.5 g-N_2O/kg-TN in the influent). However, given the concern that past sampling regimes could have underreported N_2O flux levels, it is also possible that this is excessively conservative. Clearly this value, or any other generalised emission factor, should be used for illustrative purposes only.

Direct emission of N_2O – biosolids application to farm soils

From the operations data collected for this study, 20-40% of the sewage nitrogen ends up in the biosolids waste stream. Disposal of these biosolids therefore represents the second largest flux of nitrogen leaving the sewage treatment system, and could potentially be another substantial pathway for N_2O generation.

The available literature indicates large variability in the possibility of N_2O generation from land-applied biosolids. Recommendations for the Canadian wastewater industry ranged from 8 to 36 g-N_2O per kg-N applied, depending on the soil type of the field to which the biosolids is applied (Brown et al. 2010). No Australian studies were found that measure actual N_2O generation from fields to which biosolids have been applied. However, local SEQ studies did identify that mineralisation rates of biosolids-N were much higher than is assumed by guidelines for calculating biosolids application rates (Barry and Bell 2006, Pu et al. 2008). This implies a higher risk of N_2O emissions than might otherwise be expected by the industry.

Biosolids use on agricultural fields can reduce the need for alternative fertiliser products, which themselves would have otherwise caused some degree of 'anthropogenic' N_2O emissions. As with biosolids use, the actual N_2O generation from synthetic fertiliser use could vary greatly depending on crop type, the choice of management practices, and climatic factors. In the Australian context, the potential for soil-generated N_2O emissions is considered to increase in warm or humid climates, and where crop irrigation is practiced (Thorburn et al. 2013). A recent review of Australian field data indicates that agricultural N_2O emissions can vary from less than 0.2% of applied N (substantially lower than the default IPCC value), to as high as 27% of applied N, depending on the crop type and location (Thorburn et al. 2013).

The case study analysis does not utilise literature based emission factors for field N_2O fluxes, because of (a) the lack of locally relevant guidelines that accounts for mineralisation rates of the (predominantly) organic nitrogen in biosolids; and (b) the difficulty in achieving an 'even' handling of the uncertainty across the two different fertiliser types.

Emission factors were instead taken from the IPCC guidelines for agricultural inventory (2006), which recommend using the same value (15.7 g-N_2O per kg-N applied to soils) for both mineral and organic nitrogenous fertilisers. The net N_2O flux attributed to the biosolids application therefore becomes dependent on the assumption used for bioavailability of the biosolids-N, which dictates the ratio of total biosolids-N (applied) to total synthetic fertiliser-N (avoided).

Clearly there is potential for large spatial and temporal variability in the net-N_2O flux attributed to the application of biosolids to farming soils. While conventional N_2O emission factors have been used in this study, these should be considered as highly uncertain and therefore most useful if taken as an indicative guide.

4.2 Assessment of Life-cycle Impact Assessment models

Ozone Layer Depletion

> *This section draws on the literature review in the manuscript:*
>
> *(Appendix A1): Lane, J. (in press) Stratospheric Ozone Depletion. Chapter 5 in 'Compendium of Life Cycle Assessment'. (Eds W. Kloepfler and M.Curran) Vol.4, Life Cycle Impact Assessment – LCIA, Springer.*

The stratospheric ozone layer plays a critical role in regulating conditions for life on Earth. Anthropogenic halocarbon emissions have substantially depleted ozone levels in some parts of the stratosphere, resulting in increased transmission of UVB radiation to the surface (Rowland 2006, WMO 2011). Elevated exposure to UVB radiation has been implicated in a range of negative human and ecosystem health impacts (Lucas et al. 2008, UNEP 2010, UNEP EEAP 2012).

The 1987 Montreal Protocol delivered widespread reduction in anthropogenic, harmful halocarbon emissions. This has been credited with preventing the complete collapse of the stratospheric ozone layer (Garcia et al. 2012, Newman et al. 2009), avoiding substantial increases in surface UVB levels (Newman and McKenzie 2011) and human health effects (Newman and McKenzie 2011, van Dijk et al. 2013).

With the ozone layer beginning to recover, scientific attention is now turning more to the question of longer term ozone layer management. Growing anthropogenic emissions of N_2O are now the biggest threat to ozone layer recovery (Daniel et al. 2010, Ravishankara et al. 2009), although this risk is more than offset by the mitigating effects associated with growing emissions of CH_4 and CO_2 (Fleming et al. 2011, Plummer et al. 2010, Revell et al. 2012). The result is expected to be an ozone layer 'super recovery' - whereby global ozone abundance greatly exceeds the levels found in pre-industrial times, albeit distributed spatially in a very different way (Austin et al. 2010, Eyring et al. 2010, Fleming et al. 2011, Oman et al. 2010, Plummer et al. 2010). This changing balance is expected

to increase the gradient of surface UVB radiation levels across the world, with elevated levels persisting in the tropics, and depressed UVB radiation levels persisting in other regions (McKenzie et al. 2011). The latter situation might increase the incidence of diseases associated with insufficient UVB exposure (Freedman 2008, Lucas et al. 2006, Norval et al. 2011, UNEP EEAP 2012).

While it is unclear how the international policy community will respond to these new challenges, it seems likely that the interface of ozone layer science and management will become more complex than in the past. It may be that the metrics used for ozone layer analysis will also need to evolve, if LCA is to remain relevant to this new management paradigm.

For as long as steady-state *Ozone Depletion Potential* factors remain the conventional approach for midpoint LCA, the inclusion of N_2O is relatively straightforward. Three separate modelling studies have produced consistent steady-state ODP factors of 0.017 kg-CFC11/kg-N_2O (Ravishankara et al. 2009), 0.018 kg-CFC11/kg-N_2O (Fleming et al. 2011) and 0.019 kg-CFC11/kg-N_2O (Daniel et al. 2010), all based on year 2000 atmospheric conditions. An average of these (0.018 kg-CFC11/kg-N_2O) is used for all case study analysis in this thesis.

While this may provide useful insight to the urban water industry about the potential significance of any push to regulate N_2O for ozone layer protection (Kanter et al. 2013), it also risks the introduction of substantial uncertainties into the *OD* metric. The atmospheric modelling community has been cautious in their recommendation of the available ODP factors for non-halocarbons, because their effects on the ozone layer are manifested in very different ways to those of halocarbons. This disparity increases the chance that substance comparisons could be unevenly affected by inherent bias in the underpinning atmospheric models (Daniel et al. 2011; Fleming et al. 2011).

The way forward for inclusion of CH_4 and CO_2 into ozone layer assessment metrics is more complex, and considered too problematic for inclusion in this thesis. ODP factors have been published for both substances (Fleming et al. 2011), although it is questionable whether they could be meaningfully integrated into a holistic metric. Anthropogenic emission of CH_4 and CO_2 gases will influence the state of the ozone layer by more varied, and less direct, pathways than do halocarbons and N_2O (Fleming et al. 2011, Forster et al. 2011, Plummer et al. 2010, Portmann et al. 2012, Revell et al. 2012, Rowland 2006). Furthermore, as the drivers of the future 'super recovery' in stratospheric ozone abundance, they might be considered as the prime 'culprits' in any increased incidence of human health and ecosystem effects relating to surface UVB deficiency (as a result of excess stratospheric ozone). Combining these effects with those more conventionally associated with ozone layer management might require a holistic redesign of the LCA metrics for assessing ozone-affecting substances.

Minerals resource depletion

> *This section draws on the literature review in the manuscripts:*
>
> *(Appendix B2): Lane, J.L., de Haas, D.W., Lant, P. (under review) The diverse environmental burden of city-scale urban water systems – Supplementary Information. Submitted to Water Research.*
>
> *(Appendix C1): Lane, J., Lant, P. (in prep) Biosolids and agriculture – a growing 'LCA risk' for urban water utilities.*

A diverse range of models exist for assessing the sustainability implications of resource depletion (or recovery) in LCA, without any consensus being reached in the LCA research community on the preferred approach (Berger and Finkbeiner 2011, Klinglmair et al. 2014, Schneider et al. 2014, Steen 2006, Stewart and Weidema 2005, Yellishetty et al. 2011). In part, this reflects the variety of fundamentals that are addressed by the different metrics. Some take a thermodynamic approach to measure intrinsic properties (e.g. Dewulf et al. 2007), but take no account of the role that resource scarcity plays in influencing sustainability concerns. Others use mass-balance based metrics to express the relative scarcity of physical stocks (Hauschild and Potting 2005, Schneider et al. 2011, van Oers et al. 2002), but have limited capacity to account for the importance of declining ore quality, nor the possibility that changes in the supply-demand balance inevitably encourage the discovery of new deposits. Other recent approaches strive for a compromise, using metrics based on changing ore body quality to incorporate both scarcity and resource quality constraints (Goedkoop et al. 2009, Swart and Dewulf 2013, Vieira et al. 2012).

Because of the different fundamentals that are employed, the relative importance of specific minerals can vary greatly depending on the chosen metric (Goedkoop et al. 2009, Klinglmair et al. 2014, Schneider et al. 2014, Swart and Dewulf 2013). It is therefore likely that studies focussed on specific minerals could reach very different conclusions, depending on the metric that is employed.

The resources assessment in this study has a particular focus on exploring the significance of recovering phosphorus from the wastewater system. Phosphorus is a critical, non-substitutable, ingredient in food production, and global demand is forecast to grow substantially over the next century (Van Vuuren et al. 2010). This is expected to drive up the price of mineral phosphorus fertilisers, increasing the financial incentives for phosphorus recovery from urban wastewater systems (Sartorius et al. 2012, Van Vuuren et al. 2010, von Horn and Sartorius 2009). While sewage phosphorus constitutes only a small portion of overall global and national-level P flows (Chowdhury et al. 2014), its recovery does offer the opportunity to buffer local regions against future phosphorus price increases (Cordell et al. 2013).

Of the many LCA models that purport to assess mineral resource sustainability, only three were found that include characterisation factors for mineral phosphorus resources, and are therefore capable of

incorporating biosolids phosphorus recovery into the overall analytical framework. One of those – the EPS2000 metric (Steen 1999) – relies on an assessment of implications in the distant future when concentrated ore bodies no longer exist. This approach appears to have gained little traction in the LCA research community. The other two – CML-IA (van Oers et al. 2002) and EDIP (Hauschild and Potting 2005) – have been widely adopted in applied LCA literature. The latter two approaches are structurally very similar, both employing simplified metrics based on ratios of usage:stocks.

Of these three options, the CML-IA metric was adopted for Case Study #1, as it incorporates the most recently updated data on minerals resource stocks. Characterisation factors were taken from version 4.2 of the CML-IA method for *Abiotic Depletion Potential* (CML 2013). That *Abiotic Depletion Potential* model integrates both mineral and fossil fuel resources (see van Oers et al. 2002), however the latter were not used in this study. The issue of fossil fuel resource depletion is captured here under a separate impact assessment category.

The CML-IA methodology for *Abiotic Depletion Potential* includes three possible metric alternatives, with each essentially addressing a different timeframe of concern (van Oers et al. 2002). The 'ultimate reserve' approach takes the longest term view, benchmarking the depletion against estimates of the total mineral stock in the Earth's crust. At the other end of the spectrum is the 'economic reserve' approach, which benchmarks the depletion against the stock levels currently considered to be economically viable to extract. The 'economic reserve' metric is used in this analysis. Of the three CML-IA possibilities, this is the one most closely aligned with the nature of the problem pertaining to future phosphate rock availability. That judgement assumes that the most pressing risk related to phosphate resource availability is the potential for future price rises to compromise food production in poorer regions of the world. In that context, it is the shorter term supply-demand benchmarks of the 'economic reserve' metric that are of the greatest interest.

The CML-IA metric utilises data on the total known in-ground resource considered to be economically viable to mine, taken from US government estimates (Jasinski 2009). However, the general sentiment in public discourse is that public knowledge of the state of mineral phosphate reserves is rather weak. It is unclear how much this would affect the relative uncertainty associated with the CML characterisation factor specifically for mineral phosphorus.

5. RESEARCH OUTCOMES

Overview

This section summarises six of the key findings from this study, drawing across all of the publications included in the thesis appendices.

5.1 Urban water system operations contribute to a diverse set of life-cycle environmental and sustainability impacts

This section draws results and conclusions from the manuscript:

(Appendix B): Lane, J.L., de Haas, D.W., Lant, P. (under review) The diverse environmental burden of city-scale urban water systems. Submitted to Water Research.

Analysis of two large scale, integrated urban water supply and wastewater system configurations have shown that system operations dominate most of the environmental and sustainability impact results associated with the systems' life cycle. Wastewater treatment plays a significant role in the environmental burden of the overall urban water system, even when that system includes relatively energy intensive water supply technologies (Table 3).

Electricity use is an important driver of indirect environmental impacts across the life cycle, and increases substantially with the adoption of a more diverse mix of water supply technologies. These alternate water supply approaches reduce the environmental pressures associated with freshwater extraction and nutrient discharge, however this comes at the expense of substantial increases in many other life-cycle environmental impacts (Table 4).

Notwithstanding the uncertainties involved, it is clear that many of the key drivers of life-cycle impacts are directly within the sphere of influence of the water system managers. While electricity use strongly influenced the comparison across different water supply systems, the results suggest that it will not make a good proxy for the GHG burden of the overall urban water system. Fugitive CH_4 and N_2O emissions from dams and wastewater systems are also important in this regard, collectively contributing up to 20% of the total GHG footprint (Table 3).

Direct ecosystem pressure associated with water use and nutrient discharge are important, even when considered from the broader life-cycle perspective. Direct contributions to stratospheric ozone depletion (via N2O emissions) and ecotoxicity impacts (via sewage micropollutants) also appear substantial from a life-cycle perspective, contributing up to 90% of the overall impacts for those categories (Table 3). Note, however, that both the latter conclusions require careful attention to the quality of the impact modelling approach that is adopted (see below).

Table 3: Impact results for the Traditional & Diversified scenarios, separated into contributions from direct process flows vs. those associated with the supply chain of operational inputs. Contributions are only shown if they are ≥1% in a positive or negative direction. WW = wastewater. The impact categories shown are: Freshwater Extraction Stress (FES); Marine (MEU) & Freshwater (FEU) Eutrophication; Marine (MTX), Freshwater (FTX) and Terrestrial (TTX) Ecotoxicity; Terrestrial Acidification (TA); Global Warming (GW); Ozone Layer Depletion (OD); Fossil Fuels (FFD) and Minerals (MD) resource depletion; Photochemical Ozone (POF) & Particulates (PF) Formation – see also Table 2.

		FES (%)	MEU (%)	FEU (%)	MTX (%)	FTX (%)	TTX (%)	TA (%)	GW (%)	OD (%)	FFD (%)	MD (%)	HTX (%)	POF (%)	PF (%)
Direct	dam extractions	104/101													
	WW to waterways[a]		96 / 94	0 / 1	81/76	0 / 19							3 / 3		
	WW to irrigation[a]	-6 / -6				5 / 4	7 / 7				-1 / 0		3 / 4	-1 / 0	
	biosolids transport + use[a]		3 / 5	201/196	2 / 2	40 / 31	92 / 92					0 / 0	61 / 51		
	avoided fertiliser use[a]		-2 / -3	-106/-104		-1 / 0	-1 / -1					0 / 0	-28 / -24		
	Dams - CH_4								4 / 1						
	WW system - CH_4[b]								6 / 3					1 / 0	
	WW system - CO_2[b]								3 / 2						
	WW system - N_2O[b]								7 / 4	80 / 70					
	WW system - NH_3[b]		1 / 2					41 / 27	1 / 0	6 / 5					23 / 14
	WW system – other							1 / 1						1 / 1	1 / 1
	WW system -C sequest'n[b]								-3 / -2						
Inputs	electricity use		1 / 3	1 / 2	2 / 6	6 / 11	0 / 1	41 / 61	59 / 77	12 / 23	69 / 82	22 / 42	11 / 22	70 / 83	55 / 73
	waste transport				6 / 6	2 / 2		1 / 1	2 / 1		4 / 2		16 / 14	3 / 1	2 / 1
	chemicals supply	0 / 1		3 / 3	2 / 3	32 / 24		8 / 5	7 / 4		8 / 4	22 / 18	7 / 7	7 / 4	8 / 5
Construction materials		1 / 3		2 / 1	6 / 6	15 / 9	1 / 1	8 / 6	13 / 8	2 / 2	19 / 12	56 / 40	26 / 21	19 / 11	11 / 7
other						0 / 1			1 / 1				0 / 1		
Total		100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%

[a] *excluding direct greenhouse & NH_3 gas emissions from the disposal site*

[b] *including direct greenhouse & NH_3 gas emissions from the disposal of wastewater and biosolids*

Table 4: Primary contributions to the difference in per-household results between the 'Traditional infrastructure' (T) and 'Diversified infrastructure' (D) scenarios. For each of the items shown, the contribution was calculated as (D-T)/ ΣT. This represents the change in impact result (D-T) for each of the items, divided by the total impact (ΣT) calculated for the 'Traditional infrastructure' scenario. Contributions to the overall change are only shown if ≥1% in a positive or negative direction. The impact categories (column headings) are defined above (in Table 3) and also in Table 2.

(b)		FES	MEP	FEP	MTX	FTX	TTX	TA	GW	OD	FFD	MD	HTX	POF	PF
Direct	dam extractions	-66%													
	WW --> water[a]		-28%	+1%	-12%	+25%							+1%		
	WW --> irrigation[a]	+4%											+2%		
	biosolids transport + use[a]			+4%											
	avoided fertiliser use[a]			-2%											
	direct gases – dams								-2%						
	direct gases - WW system[b]														
Inputs	electricity use		+1%	+1%	+3%	+8%		+52%	+75%	+15%	+88%	+28%	+15%	+89%	+70%
	waste transport														
	chemicals supply												+2%		
Construction materials							-3%					+5%	-10%	-1%	+3%
other							+1%	+1%	+2%		+1%				
Total			-62%	-27%	+4%	-9%	+31%	+1%	+54%	+73%	+16%	+93%	+17%	+19%	+92%

[a] *excluding direct greenhouse & NH_3 gas emissions from the disposal site*

[b] *including direct greenhouse & NH_3 gas emissions from the disposal of wastewater and biosolids*

5.2 Substantial uncertainties in modelling urban water system operations will impede options analysis

This section draws results and conclusions from the manuscripts:

(Appendix B1): Lane, J.L., de Haas, D.W., Lant, P. (under review) The diverse environmental burden of city-scale urban water systems. Submitted to Water Research.

(Appendix C1): Lane, J.L., Lant, P. (in prep) Biosolids and agriculture – a growing 'LCA risk' for urban water utilities.

Given the number and complexity of issues that are important to the LCA results, future urban water systems analysis will require substantial rigour in the estimation of key inventory data. Two challenges prominent in this study are the estimation of (i) nutrient losses from biosolids application to farms; and (ii) direct GHG fluxes from all components of the urban water system.

Estimating nutrient flux to waterways from biosolids use on farms

The available Australian research does highlight the potential for nitrate and phosphate leaching from biosolids application to be higher than from the use of conventional fertilisers, even at 1 NLBAR[2] application rates (Ives et al. 2011, Pritchard et al. 2007, Pu et al. 2008). However there is no evidence to suggest this would eventuate if biosolids applications were managed appropriately, nor any to justify the large loss rates that are built in to the conventional LCA-based approach used in this study. The latter relies on European based assumptions for leaching fluxes, which are generally considered to be much higher than the equivalent losses in Australia.

For analysis of biosolids disposal options in the context of Australian farming practices, it would be preferable to use more focussed information for estimating nutrient losses from different fertiliser types. This would also need to account for the influence of varying soil type, climate and farm management practices. Unfortunately, to our knowledge, there is little published material available to guide such an assessment in a standardised manner.

Estimating direct greenhouse gas fluxes

Despite substantial recent effort in characterising direct GHG fluxes associated with urban water system operations, estimation of all the (potentially) important sources remains extremely problematic. A review of the most recent literature reveals that major gaps or uncertainties constrain the quality of estimates in all cases, even for the Gold Coast urban water system where there is an unusually large amount of local empirical data. Using the best available estimates generated for this study, direct (Scope 1) emissions contribute 21% of the overall GHG footprint for an urban water system with a traditional, low energy water supply configuration (Table 3). While that contribution

[2] 'NLBAR' = Nitrogen Limited Biosolids Application Rate

falls to 10% when an extremely energy-intensive water system is employed, this remains substantial given some of the most important GHG flux estimates could conceivably be an order of magnitude higher.

Closer analysis of the biosolids system reveals that, for the option of biosolids application to farmland, field fluxes of carbon, CH_4 and N_2O are likely to be as or more important than the implications of biosolids transport, or nitrogenous fertiliser manufacture (Table 5). It is the latter two which historically have dominated debate on the merits of biosolids recovery on farms, and the uncertainties in the life-cycle environmental implications of doing so. While the Australian water industry is now supporting a more holistic consideration of GHG analysis, it remains selectively focussed on the potential for soil carbon sequestration. The results presented in Table 5 suggest this might be too limited a perspective, given the field N_2O emissions are of a similar scale to that for carbon sequestration. Importantly, the latter may well be an upper bound for what is achievable under Australian conditions, whereas the available Australian data indicates that field N_2O emissions could potentially be an order of magnitude higher than the estimate produced here.

Table 5: A breakdown of the greenhouse gas balance for the agricultural-use biosolids disposal pathway, excluding any treatment processing required onsite at the STP. Positive values indicate a GHG flux to atmosphere. Negative values indicate an offset that can be attributed to the GHG footprint – either the sequestration of carbon, or the displacement of some activity that would otherwise lead to GHG emissions. The large number of potentially substantial contributions (positive and negative) indicate that GHG footprinting for biosolids disposal on farms must pay close attention to the choice of system boundary and the uncertainty of the different estimates used.

		biosolids use	avoided fertiliser use	avoided crop production
field fluxes	N_2O	408	-220	-12
	NH_3	71	-19	-1
	CH_4 (biogenic & non-biogenic)	68		
	CO_2 (non-biogenic)	96		
	C sequestered (biogenic)	-274		
biosolids transport + application		153		
production + supply			-134	-10
other		4		
total		**527**	**-373**	**-24**
		130 kg CO2e / t-ds		

5.3 The urban water industry could be affected by future shifts in international policy on stratospheric ozone depletion

This section draws results and conclusions from the manuscripts:

(Appendix A1): Lane, J. (in press) Stratospheric Ozone Depletion. Chapter 5 in 'Compendium of Life Cycle Assessment'. (Eds W Kloepffer and M Curran) Vol.4, Life Cycle Impact Assessment - LCIA. Springer.

(Appendix A2): Lane, J., Lant, P. (2012) Including N2O in ozone depletion models for LCA. The International Journal of Life Cycle Assessment 17, 252-257.

Despite the recent scientific consensus that anthropogenic emission of non-halocarbon substances (N_2O, CH_4, CO_2) will be the strongest driver of future ozone layer condition, it remains unclear whether and how the international policy community might respond. Perhaps this explains the apparent oversight of this issue in the broader LCA community, where the notion of 'best practice' still is focussed exclusively on the assessment of halocarbons.

However, in the case of N_2O emissions at least, it is recommended that the urban water industry expand its consideration beyond that norm. Not doing so would compromise two of the fundamental benefits that LCA can offer. The first being that it provides quantitative analysis across a broad spectrum of environmental concerns, thereby providing a robust framework for including as many environmentally-relevant issues as possible into the decision making process. Secondly, it does this using best-estimate models of marginal impact that aspire to capture as much scientific rigour as is possible in each case. If LCA practitioners choose (for good reason or bad) to consider midpoint *OD* results, then it would seem beneficial to account for N_2O as a significant marginal risk under current circumstances.

The significance of taking this step is illustrated in a simple case study comparing two technology options for an urban STP. For an STP with marine discharge of treated effluent containing a total nitrogen (TN) concentration of 10mg/L, the default *OD* result ranks 16th out of 18 impact categories, and is 3-4 orders of magnitude lower than those for *Marine Eutrophication (MEU)* and *Global Warming (GW)* (Figure 3a). Notwithstanding concerns about hidden bias in drawing such a conclusion (Heijungs et al. 2007), this result implies that the STP contribution to the global ozone depletion 'problem' is trivial compared to its contributions to other environmental challenges. However if the proposed ODP factor for N_2O is included in the assessment, then the STP's contribution to global *OD* ranks 4th and is on a par with those emerging issues (e.g. global warming, ecotoxicity) already of some concern to urban water system planners.

Figure 3b then illustrates how the inclusion of N_2O in ozone depletion considerations could affect LCIA based decision making when options are being compared. The context here is the inexorable push to improve waterway health in Australian urban areas by reducing STP effluent nutrient levels.

Environmental debate on such a decision would typically extend no further than the consideration of GHG implications. For the chosen technologies, the 50% reduction in effluent total nitrogen increases the estimated life-cycle N_2O emissions by 19%, however this has little impact on the *GW* results because of significant other contributions associated with power use and CH_4 recovery. This result would suggest only a negligible tradeoff, and therefore great benefits in moving to the lower effluent nitrogen concentration. Consideration of the default *OD* results would do nothing to change this conclusion, whereas the inclusion of an ODP factor for N_2O may would not only change the ranking of the two scenarios in terms of *OD*, but also clearly identify a potentially significant downside to the advanced nutrient removal.

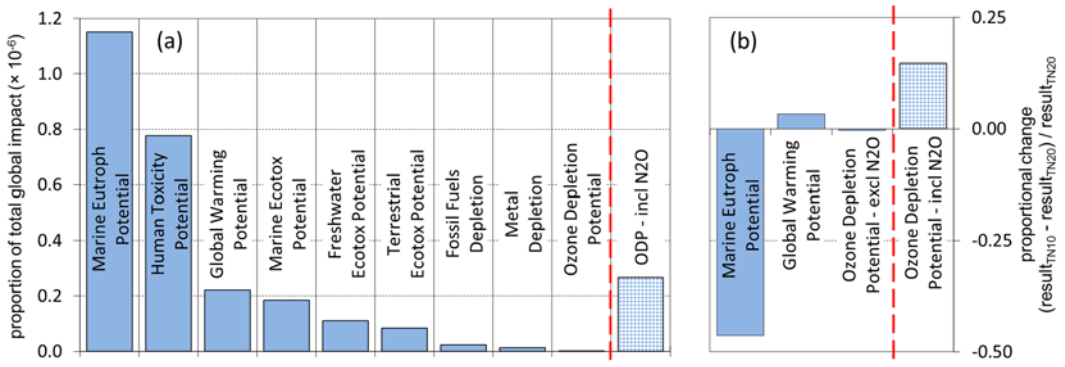

Figure 3: Selected midpoint LCIA results applying the ReCiPe midpoint (H) method to STP scenarios 5 & 6 from Foley et al (2010). The OD results are compared with/without the inclusion of an ODP factor for N2O (0.018 kg-CFC11e/kg-N2O): in (a), on an absolute basis for a scenario with total nitrogen (TN)=10mg/L, normalised against global impact estimates; and in (b), as a proportional change caused by a reduction in effluent TN from 20mg/L to 10mg/L.

While real life decision making would of course be more complex than this simplified example, it does illustrate that ozone depletion results could possibly affect planning outcomes when there are substantial N_2O emissions involved. This highlights just how easily the urban water industry could be affected if non-halocarbon emissions begin to feature more prominently in future ozone management policy.

5.4 Further development of resource depletion models is required, if LCA is to inform the debate on sewage phosphorus recovery

> *This section draws results and conclusions from the manuscript:*
>
> *(Appendix C1): Lane, J.L., Lant, P. (in prep) Biosolids and agriculture – a growing 'LCA risk' for urban water utilities.*

The two commonly used metrics[3] that include phosphate resources in their scope both utilise a similar calculation approach, meaning they can offer little insight into whether the choice of metric will affect

[3] These two are the *Abiotic Depletion Potential* impact category developed by CML van Oers, L., Koning, A.d., Guinee, J.B. and Huppes, G. (2002) Abiotic resource depletion in LCA, Directoraat-General Rijkswaterstaat., and the *Resources* impact category from the EDIP method for LCIA Hauschild, M. and Potting, J. (2005)

conclusions on the relative importance of phosphorus use relative to the depletion of other minerals. To address this gap, a characterisation factor for phosphate rock resources was developed in this thesis, in a manner consistent with the ReCiPe metric (Goedkoop et al. 2009) for assessing *Minerals Depletion*. The ReCiPe metric is one of a group of contemporary approaches to resource depletion assessment that attempt to incorporate economic principles into the calculation structure.

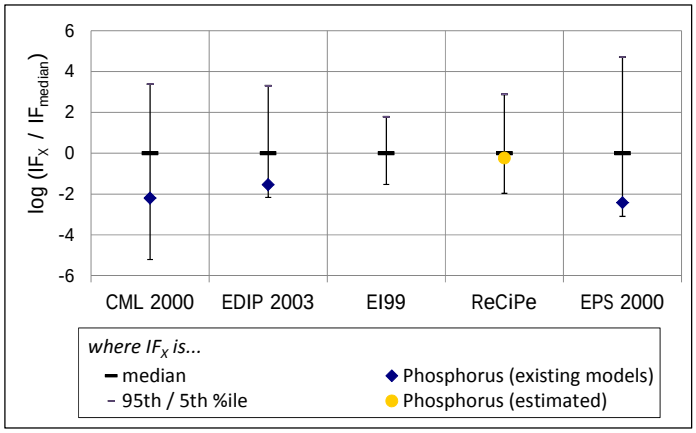

Figure 4: A comparison of characterisation factor variability across a selection of LCIA models for minerals depletion, including the three available models that account for mineral phosphate resources - CML (van Oers et al. 2002), EDIP (Hauschild and Potting 2005) and EPS (Steen 1999); and the ReCiPe-based model (Goedkoop et al. 2009) amended with a characterisation factor (developed in this thesis) for phosphate rock. The spread shown for each model indicates the extent of variation in characterisation factors across all the minerals included in that model. In all cases, Phosphorus depletion is considered less significant than the median substance on a unit-mass basis. However, the relative importance of phosphorus under the ReCiPe metric (close to the median) is much greater than under the alternative metrics (~2 orders less significant than the median).

Comparing the spread of midpoint-level impact factors illustrates the significance of choosing different LCA metrics for assessing minerals resource sustainability (Figure 4). The ReCiPe-based characterisation factor for phosphate rock is slightly less than the median of all characterisation factors in the ReCiPe *Minerals Depletion* model. In contrast, phosphorus resources are considered two orders of magnitude less significant than the median mineral under the CML metric and the other two existing models that account for phosphorus resources. In other words, using those alternate *Minerals Depletion* metrics would make the phosphorus recovery seem *less* important than if the modified ReCiPe model is used.

At face value, the consistently low (less than median) characterisation factors for phosphorus might suggest that phosphorus resource security might not warrant any special priority in broader debates on resource sustainability. This does not accord with the water industry's promotion of its objectives to increase sewage phosphorus recovery rates, nor the more general push for addressing the sustainability of global phosphorus resources. However, the comparison presented here gives no

Spatial differentiation in Life Cycle impact assessment - The EDIP2003 methodology, Danish Ministry of the Environment.. The former is used in analysis of Case Study #1 in this thesis.

indication of whether or not a similar conclusion would be reached, should phosphorus characterisation factors be available for some of the quite different metrics that have more recently become available.

Furthermore, consideration should be given to whether LCA resource depletion metrics can adequately represent the somewhat unique socio-economic characteristics of the phosphorus resource sustainability challenge. Because of the fundamental and irreplaceable role of phosphorus in food production, concerns over maintaining equity of access (across poorer countries and/or demographics) feature strongly in debates on this topic (Cordell et al. 2009). There is little, if any, precedent for including such social objectives in metrics developed for LCA resource assessment.

Further investigation into this topic will be required, if the water industry is to develop metrics suitable for assessing the sustainability implications of phosphorus recovery.

5.5 Benchmarking data provides useful perspective for more focussed options analysis

This section draws results and conclusions from the manuscripts:

(Appendix C1): Lane, J.L., Lant, P. (in prep) Biosolids and agriculture – a growing 'LCA risk' for urban water utilities.

Nutrient recovery features prominently in public discussion on the benefits of recovering biosolids for use on farms. The opportunity for this to reduce the need for energy intensive manufacture of nitrogenous fertiliser has long been recognised. More recently, the debate has extended to the role this might play in terms of helping to alleviate future shortages of mineral phosphate resources.

However, when the implications of this biosolids recovery are compared against other opportunities to reduce the energy and minerals footprints of the urban water system, the strength of this justification would appear much weaker (Table 6).

Whether using *Cumulative Energy Demand (CED)* or *Fossil Fuel Depletion (FFD)* as the proxy, it is clear there may be far easier ways for the water industry to reduce its contribution to global energy demand. The benefit of avoiding the synthetic fertiliser supply (-26 TJ; -612 t oil-e/y) is minor compared to the overall burdens associated with operation of the wastewater and water supply systems. It would seem likely that the urban water industry could easily achieve similar benefits through minor improvements to the energy efficiency of the existing infrastructure stock.

The analysis in this study also provides only qualified support for the notion that phosphorus recovery would provide substantial benefits in terms of future resource availability. In Table 3, the Minerals Depletion benefits of the avoided fertiliser supply is negligible, when compared to the minerals footprint embedded in the materials used to construct the urban water system infrastructure. The

significance of the phosphorus recovery does appear more substantial when a very different metric is used for assessing the significance of Minerals Depletion (Table 7). However, the benefits still appear relatively small unless a very high level of phosphorus fertiliser displacement can be achieved.

In this latter (Minerals Depletion) case, caution is required when interpreting these results. As discussed below, the utility of these metrics to adequately represent the issues of relevance to the phosphorus sustainability debate is questionable. Nonetheless, the results do suggest that, on some measures, the water industry might just as easily make a positive contribution to easing pressure on mineral resource sustainability by reducing or substituting other inputs to the urban water system.

Table 6: Impacts of energy use and greenhouse gas emissions associated with biosolids used as a farm soil supplement. Impacts shown are CED=Cumulative Energy Demand; FFD=Fossil Fuel Depletion; GW =Global Warming; OD=Ozone Depletion. The impacts related to biosolids disposal are benchmarked against the total equivalent impacts associated with operation of the wastewater system, and combined water supply-wastewater systems, modelled in Figure 1a.

	CED (TJ)	FFD (t oil-e/y)	GW (kt CO2-e/y)	OD (kg CFC11-e/y)
Biosolids - overall	**-2**	**-49**	**1.4**	**147**
transport to disposal site	23	547	1.6	0.5
onsite - energy use	1	25	0.1	0.04
onsite - gas emissions	-0.1	-3	4.4	156
onsite - C sequestration	0	0	-3.0	0
onsite - other	0	0	0.03	2.0
offsets - fertiliser supply to site	-26	-612	-1.5	-3.0
offsets - crop production	-0.3	-6	-0.2	-8.2
WW system - overall	**684**	**12,059**	**64**	**554**
(% contribution from biosolids disposal)	*-0.3%*	*-0.4%*	*2.2%*	*27%*
Urban water system - overall	**961**	**17,392**	**89**	**574**
(% contribution from biosolids disposal)	*-0.2%*	*-0.3%*	*1.6%*	*26%*

The example comparisons provided here illustrate the benefits of using broader benchmarks to help interpret the significance of the results from more focussed options analysis. Doing so can reveal whether or not the perceived benefits of some favoured approach really provide the most effective way for a water utility to achieve its end goals. More generally, the availability and use of suitable benchmark datasets will greatly enhance the ability for decision makers to understand and prioritise across tradeoffs, particularly when the use of the LCA methodology introduces a number of impact categories that are unfamiliar to most in the urban water industry.

Table 7: Minerals Depletion results for the overall urban water system, using the customised version of the ReCiPe Minerals Depletion metric developed for this study. Results are shown for a somewhat extreme range (low/medium/high) of assumptions that dictate the amount of phosphate fertiliser that is displaced by the use of biosolids (no other parameters were changed). According to this metric, the recovery of biosolids phosphorus has a notable, but not dominant, effect on the overall minerals resource sustainability results for the urban water system.

		Minerals Depletion (t Fe-e/y)		
		low P displacement	default P displacement	high P displacement
displaced fertiliser - biosolids use on farms		-458	-1833	-3273
total for rest of system		+1142	+1142	+1142
displaced fertiliser - urban WW irrigation		*-143*		
operational inputs for the urban water system	*chemicals*	*+153*		
	other	*+38*		
construction of the urban water system infrastructure	*networks*	*+39*		
	other	*+1055*		
urban water system - overall		+684	-691	-2131

5.6 Impact model choice has a strong influence on interpretation of urban water systems analysis.

This section draws results and conclusions from the manuscripts:

(Appendix B1-B2): Lane, J.L., de Haas, D.W., Lant, P. (under review) The diverse environmental burden of city-scale urban water systems. Submitted to Water Research.

(Appendix C1): Lane, J.L., Lant, P. (in prep) Biosolids and agriculture – a growing 'LCA risk' for urban water utilities.

In the available literature on LCA analysis of urban water technologies and systems, relatively few studies have considered whether the choice of impact assessment models will affect the conclusions that might be drawn. In those that have, the discussion is mostly focussed on the implications of choosing between different 'off-the-shelf' packages[4] of impact assessment models (Corominas et al. 2013, Hospido et al. 2012, Igos et al. 2013, Renou et al. 2008, Uche et al. 2013, Zhou et al. 2011). However, there are many reasons why these past comparisons are unlikely to provide a useful guide to urban water planners. The substantial developments in LCA impact assessment methodology over recent years (see Finnveden et al. 2009, Hauschild et al. 2013, Rack et al. 2013, Zamagni et al. 2012) have led to major improvements in model fidelity for some impact categories, but much less for others. Furthermore, these changes have been implemented inconsistently in those 'off-the-shelf' model packages, and many of the recent model improvements have not yet been analysed in terms of their significance for urban water systems analysis.

[4] In the LCA literature, these packages are more commonly termed 'impact assessment methods'.

This thesis focussed in particular on three sets of impact categories. In each case, closer scrutiny indicates that model choice can have a strong influence on the conclusions drawn from the analysis, not necessarily in a manner that is consistent with contemporary priorities in the relevant science disciplines.

Impact categories where the choice of model dictates the outcomes

As illustrated in the previous section, *Minerals Depletion* assessment is a pertinent example where one metric can give starkly different impressions than another. Given the substantial variation in the ranking of substances across the available metrics, it is clear that metric choice could strongly influence the conclusions that might be reached from analysis in other contexts than just phosphorus recovery. This challenge is exacerbated by the lack of consensus in the LCA research community on the best way to characterise this particular sustainability issue.

Ozone Depletion (OD) assessment is another case where the choice of impact modelling approach could have a significant bearing on interpretation of urban water systems analysis. Using a conventional metric gives the impression that water system managers have relatively little control over their contribution to *OD*, as the other life-cycle emissions of ozone depleting substances are dispersed across various supply chain activities (Table 8). But factoring in the role of N_2O leads to the opposite conclusion, since direct N_2O emissions from the wastewater system dominate the life-cycle *OD* results. The utility of this latter approach is discussed further in the next section.

Table 8: The effect of impact assessment model choice on OD results for the 'Traditional infrastructure' scenario. The first set of results employs a conventional LCA impact assessment model for ozone depletion, including only halocarbon emissions in the analysis. The second set of results incorporates the additional characterisation factor for anthropogenic N2O emissions developed for this study.

		excluding N2O	including N2O
Total OD (kg-CFC11-eq/y)		1.0	578
contributions	Wastewater system N2O	--	86%
	electricity generation	1%	12%
	chemicals manufacture	53%	1.0%
	transport (biosolids, chemicals)	0%	0.2%
	infrastructure construction	46%	1.8%

Impact categories that lack sufficient fidelity

Difficulties are also apparent in the use of LCA toxicity models for analysis of urban water systems. In this case, the challenge has little to with choice between competing modelling approaches, since this is one area where the LCA community has moved strongly towards the adoption of a consistent approach.

Table 9: 'Traditional infrastructure' scenario results for (a) Marine, (b) Freshwater, and (c) Terrestrial Ecotoxicity, showing the relative contributions (in %) from process streams vs. 'indirect' impacts, and metals vs. organics.

(a)	wastewater disposal	biosolids disposal	supply chain activities	Total
Metals	75	1.3	23	98
Organics	1.3	0.03	0.01	1.3
Total	76	1.3	23	100

(b)	wastewater disposal	biosolids disposal	supply chain activities	Total
Metals	2	38	54	95
Organics	3	0.07	2	5
Total	5	38	57	100

(c)	wastewater disposal	biosolids disposal	supply chain activities	Total
Metals	2	91	1	95
Organics	5	0.20	0.29	5
Total	7	91	2	100

Certainly the scope of these toxicity models aligns well with urban water industry concerns on organic and metal residuals in wastewater and biosolids streams. Direct process flows made substantial contributions to the life cycle *Marine*, *Freshwater* and *Terrestrial Ecotoxicity* of both case studies considered here. In all cases the major process contributions come from residual metals in treated wastewater and sewage sludge streams, with the equivalent contributions from residual organic micropollutants (e.g. pharmaceuticals and pesticides) being negligible. Table 9 illustrates this metals 'dominance' in the results for the 'Traditional infrastructure' urban water scenario. This bias towards the significance of metals is in stark contrast with the high priority given to research on organics in wastewater and biosolids in recent years.

A more detailed breakdown of the *Terrestrial Ecotoxicity (TET)* results indicates just how misaligned the LCA model is with the current state of scientific thinking. Trace metals in the biosolids, particularly Copper (Cu), Zinc (Zn) and Mercury (Hg), are the dominant cause of the *TET* as measured using the LCA approach, with the toxicity offsets from avoiding heavy metals in DAP fertiliser being two orders of magnitude lower (Table 10). This conclusion does not match those from recent Australian research, finding that the copper and zinc in biosolids pose little toxicological threat to Australian soils if applied at typical rates (Pritchard et al. 2010). Furthermore, it is inconsistent with the argument that alternative phosphorus sources are needed to avoid growth in fertiliser heavy metal contamination as lower grade phosphate-ore bodies are utilised into the future (Cordell et al. 2009).

The problems with relying on the readily available, and widely used, LCA toxicity models is apparent in the underlying literature. For example, recent work has highlighted that conventional LCA approaches can, under some conditions, overstate the implications of metals emissions by an order of magnitude or more (Diamond et al. 2010, Gandhi et al. 2010, Gandhi et al. 2011, Ligthart et al. 2004). Other studies have demonstrated that, because of difficulties in identifying and measuring all the relevant organic species, chemical-specific analysis (using an approach with similarities to that used in LCA) could understate the toxicity implications of residual wastewater organics by an order of magnitude or more (Escher et al. 2011).

Table 10: Terrestrial Ecotoxicity results for the agricultural use of biosolids, broken down by toxicant group and system component. Contributions less than 0.1% are not shown, to highlight how small come of the non-zero contributions are.. Note that the convention in LCA toxicity assessment is to treat contributions <1% as statistically insignificant (Rosenbaum et al. 2008).

			Cu	Zn	Hg	Se	Cd	other metals	organics	other
biosolids application	biosolids contaminants		65%	17%	5%	3%	0.1%	1%	0.1%	
	transport & application									
displaced fertiliser	fertiliser contaminants			-0.3%			-0.2%	-0.1%		
	manufacture & supply							-0.15%		
displaced crop production	fertiliser contaminants									
	other									
remainder of the urban water system		operations	0.7%	0.4%	0.3%			1.5%	4.8%	
		construction	0.1%	0.2%	0.3%			0.2%		
total			44.3 t 1,4-DCB-e/y (95%)						2.3 t 1,4-DCB-e/y (5%)	

In light of these barriers, it is worthwhile to consider the preferred role for LCA toxicity assessment models in the analytical toolkit of water system planners. The LCA approach would never be capable of replacing the more focussed, local risk assessment that is the conventional approach to water industry toxicity analysis. However, at least in concept, it is an ideal vehicle to (a) compare the scale of local impacts with those being caused elsewhere in the activities of water industry supply chains; and (b) integrate toxicity considerations into a consistent, broader analytical framework that spans a broader spectrum of environmental issues. The objective here would be to identify those occasions when an excessive focus on local toxicity risks leads to an unacceptable degree of burden shifting. Unfortunately, it would seem that this is not yet possible in the case of LCA applied to urban water industry operations.

5.7 Shortcomings in the available LCA impact assessment models pose a risk to urban water utilities reliant on biosolids application to farmland

This section draws results and conclusions from the manuscripts:

(Appendix C1): Lane, J.L., Lant, P. (in prep) Biosolids and agriculture – a growing 'LCA risk' for urban water utilities.

The analysis to this point indicates a number of challenges that may constrain the utility of LCA when applied to biosolids use on farms. Assessing the life-cycle implications of biosolids use on farms is extremely sensitive to assumptions on a number of issues, ranging from nutrient and carbon fluxes in the soil, to the likelihood that changed farm management practices can be sustained over the medium term. The lack of information available to guide such assumptions highlights that the scope of Australian research endeavour is inconsistent with the informational needs required for robust LCA analysis.

Even more important is the failure of conventional LCIA models to support the policy and scientific arguments used in favour of promoting biosolids use on Australian farms. As described above, the LCA results do not provide a strong justification for the pursuit of nutrient recovery in the name of avoiding energy intensive fertiliser manufacture, or avoiding depletion of mineral phosphate resources.

Critically, it is clear that LCA *Terrestrial Ecotoxicity* models contradict the best available Australian risk assessment, and should be excluded from analysis of biosolids disposal options. Of itself, this is not a novel conclusion – the limitations in terms of metals analysis have been recognised for some years by academic and industry analysts using LCA to focus on biosolids disposal options (e.g. Corominas et al. 2013, Hospido et al. 2005, Peters and Rowley 2009). However it appears that message has had less influence than might be expected, with water industry studies continuing to highlight the "significance" of biosolids metals using LCA toxicity models (e.g. Hospido et al. 2008, Hospido et al. 2010, Mahgoub et al. 2010, Pasqualino et al. 2009).

While it remains valid to exclude LCA toxicity (and other) impacts from biosolids analysis aimed at providing insight to the urban water industry, this may not be the case if the primary motivation for conducting LCA of biosolids use becomes the agricultural perspective. Toxicity analysis has been identified as a priority for inclusion in the expansion of LCA applied to Australian agriculture (Eady et al. 2012), reflecting the desire to incorporate agri-specific issues such as pesticides use and synthetic fertiliser contaminants, and benchmark such results against the potential impacts that occur elsewhere in the system lifecycle. Eutrophication analysis has also been raised as a priority for Australian agri-LCA, despite the lack of models sensitive enough to characterise differences in nutrient flux associated with changing fertilisation approaches.

It appears that the use of LCA to inform Australian agricultural management is set to grow, driven by the international push for greater information on the environmental attributes of different products being traded in the global marketplace. This might require a very different set of methodological approaches (system boundaries; inventory assumptions; impact categories) than have been employed in past water-industry sponsored analysis of biosolids disposal options. Some of these approaches could make biosolids reuse on farms look substantially less favourable than is indicated by current scientific wisdom – for example, the use of conventional impact assessment models might 'reveal' that biosolids use on farms has large downsides for the environmental footprint of the farm in question (Table 11). If this were to reduce farmer enthusiasm for utilising biosolids as a crop supplement, this could pose a substantial risk over the medium to longer term for those Australian water utilities already heavily reliant on the agricultural disposal option.

Table 11: Change in the overall life-cycle impacts for three benchmark systems, associated with the decision to apply biosolids to agricultural fields. The change measured is the difference in impacts allocated to biosolids disposal for the two pathways presented in Figure 2. A positive/negative result indicates that the impacts of the agricultural disposal source (as per figure 2b) are higher/lower than those for the landfill disposal scenario (as per figure 2a). The 'WW management system' and 'Urban Water System' benchmarks are those modelled for the 'Traditional infrastructure' scenario of Case Study #1 (see Figure 1 and Table 3), but with the destination of the biosolids changed to landfill (in the original Case Study #1 analysis, the biosolids are directed to agricultural application). The 'crop production' benchmark utilises the same sorghum grain inventory as for the estimation of cropping offsets.

	AOD (kt O_2-e/y)	TTX (t 1,4-DCB-e/y)	HTX (t 1,4-DCB-e/y)	MD (t Fe-e/y)	FFD (t oil-e/y)	CED (TJ)	GW (kt CO2-e/y)	OD (kg CFC11-e/y)
change in total impacts of biosolids disposal	**2.4**	**43**	**3.2**	**-1866**	**115**	**11**	**-14**	**42**
change in total impacts for operating the wastewater system	+16%	+1,231%	+162%	-2,439%	+1.0%	+1.6%	-18%	+8%
change in total impacts for operating the urban water system	+16%	+1,157%	+101%	-2,278%	+0.7%	+1.2%	-14%	+8%
change in total impacts of crop production (5 yr cycle)	+65%	+496%	+10%	-84%	+5%	+11%	-97%	+8%

6. CONCLUSIONS & RECOMMENDATIONS

Energy use and greenhouse gas footprints

Conclusions

Electricity use strongly influenced the LCA results for two city-scale, integrated urban water systems - one with a mains water system dominated by low-energy, dam-based water extractions; the other involving a 7-fold increase in the energy intensity of water supply. For a power supply mix dominated by coal-fired generation, this leads to a very substantial increase in the life-cycle burden of the urban water system, indicating there will be downsides to the goal of reducing pressure on local aquatic ecosystems caused by freshwater extraction and nutrient discharge.

However, measuring (or predicting) electricity consumption *per se* will not make a reliable proxy for the bigger picture environmental implications associated with the infrastructure of urban water utilities. The increase in power consumption is not the only important issue. Even for the high energy scenario, other aspects of the urban water system make substantial contribution to impact category results that are only weakly affected by the electricity supply chain.

This study compiles a comprehensive inventory for GHG emissions directly from urban water system operations, having access to a rare level of locally specific empirical data and expert knowledge. These direct (scope 1) sources contribute 10-21% of the overall GHG footprint, depending on the energy intensity of the water supply system in use. In neither case should this be considered a trivial amount. The largest potential sources of direct emissions - CH_4 from water supply dams; CH_4 from sewage reticulation; N_2O from the STP nutrient removal; N_2O from the application of biosolids to farms - all can vary by orders of magnitude depending on location, system configuration, and other factors. For other urban centres, the overall contribution from the direct emissions could therefore be much higher or lower than shown here. It seems likely that these direct (scope 1) emissions will add a substantial liability to the greenhouse footprint of many urban water systems in Australia and elsewhere.

The importance of this uncertainty and variability will be magnified when the analysis is focussed on a specific sub-component of the overall system, as is the more conventional context for decision making in the industry. The Australian industry's preference for biosolids disposal on farmlands is another issue where much of the traditional debate has focussed on energy use – in this case, the tradeoff between biosolids transport vs. manufacture of synthetic nitrogen fertiliser. Those two issues are important cost considerations for water utilities, and farmers, respectively. However, that cost-energy focus has limited value as a broader proxy for environmental perspectives. The uncertainty

associated with direct field C and N fluxes indicates will be more influential to the GHG footprint of biosolids use on farms, than will variations in the distance from farm to paddock.

Reflections and recommendations

Not addressed here, but apparent from a detailed review of the available literature, is the gap between the needs of quantitative LCA studies, and the scope of available research on direct GHG emissions associated with the urban water industry. Decision making usually involves a comparison of options or proposed system change, whereas the majority of available literature data quantifies the overall scale of emissions for single isolated systems. That is, of course, an important step in understanding the highly complex biological processes involved. Understandably, it is also the industry's most immediate concern in their attempt to understand the scale of potential exposure to carbon pricing mechanisms. However it would be a mistake for urban water planners to assume that this 'auditing' mentality will adequately inform the options comparisons that deliver change in the industry, and essentially dictate the GHG footprints of their future infrastructure.

For this to be addressed, continued research focus will be required on the development of deterministic models that can account for, and distinguish between, GHG generation and anthropogenic emission levels in response to system change. The scope of that research should include water supply dams and biosolids disposal methods – not just the sewage treatment plants that have dominated the focus of Australian investigations to date.

The extensive compilation of estimates provided here should make a useful starting point for revisiting some of the remaining gaps in the analytical framework. Further investigation is warranted into the following issues:

- While there is a need identified for further fundamental research into direct emission fluxes, it does not follow that this is the only high priority. In some cases there might also be substantial benefit in the expert community producing a synthesis explicitly tailored to the needs of decision makers. In the context of LCA-based options analysis, 'rough but science-based' can be intrinsically more useful than 'leave it out because the science is too uncertain'.
- There is also emerging evidence on direct emission sources not yet included in macro-level assessments, such as N_2O generation in gravity sewers (Short et al. 2014) and water supply dams (Musenze et al. under review). Neither would be expected to change the conclusions drawn in this study, but that should be confirmed in future analysis.
- In terms of electricity use, cities contemplating the large-scale rollout of dwelling-scale rainwater tanks would benefit from systematic assessment of the opportunity for reducing

their energy burden – given the seemingly substantial design inefficiencies that have become the norm in Australian implementation to date.

- The scope of this study was deliberately constrained to the water supply and wastewater management infrastructure that is the traditional domain of planning by urban water utilities. Future research should consider the life cycle impacts in the context of the stormwater management, and water end use, components of the urban water system.

The broader inventory data challenge

Conclusions

Other process flows, also difficult to quantify, are influential to the broader set of impact categories considered in LCA studies. One that received some attention in this thesis was the nutrient balance associated with biosolids application to farmland soils. This is a difficult challenge in predictive, quantitative studies, because of the many, highly uncertain, assumptions that are required. Firstly there is the calculation of displaced fertiliser quantities, which in LCA studies is most commonly based on a stoichiometric ratio of bioavailable nutrients. However, given the complexity inherent in agri-system management, and the available evidence on phosphorus accumulation in Australian soils, that seems a fairly tenuous assumption for non-expert engineers in the water industry to make. The case study results indicate that the LCA implications of biosolids application to farms can be very sensitive to assumptions made on future farmer behaviour.

This fertiliser displacement balance will not only affect the amount of fertiliser manufacturing and transport that is displaced, but also the likelihood and scale of nutrient fluxes from the paddock to air and to waterways. This issue is handled poorly in conventional LCA practice, which typically estimates these fluxes (e.g. N_2O emissions; or N and P leaching losses) as a linear function of total nutrient loadings to the soil. That approach overlooks the potential implications of key differences between biosolids and conventional fertilisers. For example, biosolids applications will involve greater short term nutrient excess, and contain substantial organic nutrient fractions that could mobilise at different rates than do mineral nutrients. Furthermore, other constituents of the biosolids (e.g. the high moisture content, copper and other minerals) will have an influence on the biochemistry of the soil.

In the Australian context, there is scant available data to guide the quantitative estimation of gaseous losses, or nutrient fluxes to waterways, when comparing synthetic and alternative fertilisers. For such location-specific issues, applying ‘conventional’ estimation approaches will be problematic.

Reflections and recommendations

While it is clear that these and other issues warrant closer scrutiny, addressing such uncertainties in LCA inventory data can add substantially to the level of effort required. As most users of LCA will already know, such effort can add tremendous value to an organisation when it encourages rigorous collection and review of data for their own operations. That was certainly the experience from the early stages of this study, when the focus was on the 'engineering' data obtained directly from the water utility involved. However, water utilities would gain little short term practical benefit from a detailed review of soil nutrient balances for biosolids applied to privately owned farms. It is therefore not surprising that, despite such a large number of water industry focussed LCA studies being conducted on biosolids disposal, so little attention has been paid to exploring the quality of assumptions relating to the interaction of biosolids with farm management systems.

This highlights the methodological conflict inherent in conducting LCA studies – the organisation undertaking the study benefits most immediately from a focus on issues that affect it directly, whereas a fundamental objective of LCA is to apply rigour across all parts of the system life cycle that might make a substantial contribution to environmental impacts. It became apparent during the course of this project that an appropriate balance had not been struck – the effort put into estimating 'engineering system' flows came at the expense of effort into other issues that could be just as important to the learnings from the study. I expect I am not the first to get this balance wrong. It is my experience, from working in industry, government and research sectors, that people commonly focus most of their analytical effort on things they know most about. Unfortunately that brings a risk of over-simplifying too many of the assumptions on 'external' factors. This could lead to predictable answers that reinforce, rather than challenge, a focus on internal issues.

The value of whole-of-system studies

Conclusions

Multi-criteria, quantitative impact assessment of whole-of-system infrastructure can provide substantial value to guide decision making and policy debate for the urban water industry. The analysis provided here indicates two important conclusions. Firstly, the wastewater management component makes the biggest contribution to the overall environmental burden of an integrated urban water system, even when that system is based on very energy intensive water supply technologies. Secondly, GHG footprinting is unlikely to provide an adequate decision proxy for addressing the broader range of impacts associated with urban water system operations. For rapidly growing urban centres, policy debate should focus as much on the broader environmental challenges associated with managing the increased sewage loads, as on the energy use of alternative water supply options. At

the time this PhD commenced, it was the latter that dominated public discourse on infrastructure options for future water cycle infrastructure in South East Queensland.

The whole-of-system results generated for this thesis were also used to benchmark those from a more focussed case study on biosolids disposal, so as to consider the arguments in favour of sewage nutrient recovery. Even when benchmarked against the low energy water system, the energy benefit of avoiding manufacture and supply of synthetic nitrogenous fertiliser was minor compared to the direct electricity use required to operate the wastewater and water supply systems. In terms of maintaining valuable mineral resource stocks for use by future generations, the importance of phosphorus recovery seemed, at best, to be no more substantial than the resources embedded in the physical infrastructure stock. In both cases, the benchmarking suggests that these environmental pressures could be alleviated just as effectively by reducing or substituting other inputs to the urban water system.

While the utility of that latter phosphorus comparison is questionable (as discussed below), the value of the analytical approach is demonstrated. Comparing case study results against macro-level benchmarks will help decision makers to prioritise across the environmental tradeoffs that inevitably arise during multi-criteria assessment processes.

Reflections and recommendations

A more general conclusion drawn from this study is that, even with the impact modelling uncertainties discussed below, it is likely that most key impact drivers are directly within the sphere of influence of urban water system managers. It is also apparent that pressures on local aquatic systems are important, even when considered from the broader life-cycle perspective. Focussed impact/risk assessment should therefore remain an important step in urban water decision making, when there are local environments bearing the brunt of water system activities.

Notwithstanding that, LCA does have an important role to play in urban water systems analysis. LCA provides a robust framework for quantitatively addressing (a) the more global of these 'direct' impacts, along with (b) those more indirect drivers where the impacts are likely to be felt at other points in time or space. This could prove increasingly useful as the urban water industry extends its focus beyond the traditional concerns of microbial human health protection, financial cost, water use and nutrient discharge.

The value, and challenges, with broad-spectrum impact assessment

Conclusions

There is a growing scientific focus on the interactions between GHG emissions and the stratospheric ozone layer, and the water industry should keep a watching brief for any international policy response to this shift in emphasis. Assessment of ozone layer depletion has a long history in LCA, however the LCA community has been slow to act on the recent attention given to non-halocarbon emissions. The impression is given that many view the ozone layer as an environmental problem that has been 'fixed'. However, it is possible that the looming 'super recovery' will lead to prolonged health risks caused in some regions by excessive UV exposure, whereas in other regions the main problem will be insufficient UV exposure. Anthropogenic N_2O, CH_4 and CO_2 emissions will have a pivotal role in determining if and how this might unfold. For LCA practice, there is not yet sufficient information available to guide a quantitative handling of ozone-enhancing effects of anthropogenic CH_4 and CO_2 emissions. However, the path forward for N_2O emissions is clear and simple. As the biggest cause of stratospheric ozone depletion, the inclusion of N_2O in LCA models for assessing ozone layer depletion is entirely consistent with the marginal impact assessment favoured by the LCA community. Doing so clearly indicates that the water industry could be affected by any policy attention on the links between N_2O and ozone layer depletion.

LCA toxicity assessment is another domain where spurious conclusions could result from a lack of attention to the fidelity of the chosen impact assessment models. With concerns growing over organic micropollutants in wastewater and biosolids discharges, the LCA methodology should in theory be a useful complement to the water industry. Using LCA to assess toxic effects associated with supply chain activities would allow decision makers to understand whether a focus on wastewater organics removal was simply transferring the 'toxicity problem' to other locations. Unfortunately, accepted 'best practice' LCA toxicity models are not able to provide meaningful comment on the relative importance of residual metals and organics in the water industry process streams. This also prevents meaningful interpretation of direct versus supply chain effects. Future urban water industry studies should give careful consideration to whether LCA-based toxicity analysis will provide results that offer meaningful insight to the industry.

For Australian water utilities reliant on biosolids disposal to farmlands, the lack of fidelity in available LCIA models could represent a business risk. The export-dependent Australian agricultural sector is one example of the growing momentum and international reach of LCA application. The agricultural industry has embarked on a process to encourage the use of LCA by Australian farmers, to satisfy growing international pressures to provide such information to their export customers. The analysis undertaken for this thesis indicates that the use of conventional 'best practice' impact assessment

models could bias against the use of biosolids as an environmentally favourable fertiliser option. These models contradict strongly the available field science findings that biosolids application to Australian soils poses little risk associated with biosolids metal toxicity. At the same time, available LCA metrics provide no support for the notion that biosolids phosphorus recovery makes a useful contribution to mitigating resource sustainability concerns. If the perspective of farmers becomes the main driver for decision making, it is likely that perspective will be influenced by externally imposed LCA frameworks. Water utilities would face significant strategic challenges if that reduces enthusiasm in Australia for the application of STP biosolids to farming lands.

Reflections and recommendations

The majority of LCA studies globally are conducted by users of LCA, rather than methodological researchers, whose LCA skills are strongest in the task of inventory collection. The experience of this PhD is that exploring the fidelity of LCA impact assessment models adds enormously to the effort involved in conducting an analysis. It is therefore easy to foresee that many practitioners might pay insufficient attention to uncertainties embedded in impact assessment models that are complex, confusing and/or confounded. Whether or not those impact categories get included or excluded in the final results is somewhat irrelevant – the value of the exercise will be diminished if the LCA results are not understood appropriately.

LCA should be able to provide unique perspectives of relevance to urban water planners wishing to chart a course towards a lower-impact industry, with the flexibility to adjust to changing 'sustainability' challenges over time. When it comes to identifying the impact assessment models needed to provide those perspectives, it will not be sufficient to uncritically choose off-the-shelf packages or follow past conventions. Practitioners using LCA to inform urban water industry decision making should resist the temptation to treat the impact assessment part of LCA as a black box that is somehow full of magic tricks.

If the Australian urban water industry is to gain maximum value from all that LCA has to offer, it will need to encourage the development of impact assessment models that have more relevance for its analytical needs. The same applies if the industry wishes to minimise risks associated with the growing use of LCA to support agricultural decision making. It would not be wise to wait for the international LCA research community to respond to this challenge. Instead, it would be far more effective to proactively engage with local experts in each relevant discipline. In my reviews of the different LCIA models and underpinning science base, I repeatedly found scientists who recognised that such collaborations would help them find ways to incorporate their own expertise into broader

decision making frameworks. As pointed out recently by Huijbregts (2014), this is a message with equal relevance to the broader LCA community operating in other industry sectors.

In the meantime, the urban water industry in Australia and overseas would benefit from a more comprehensive review of the influence that LCIA model choice can have on the conclusions drawn from analysis of contemporary management options.

In terms of impact modelling improvements, specific recommendations arising from this thesis include:

- If LCA analysis of urban water systems are to include ozone depletion as one of the impact categories under consideration, then N_2O should be included in the analytical scope. Global warming footprints will not be an adequate proxy to capture this issue in multi-criteria options comparisons.

- Four investigations are required to enable informative LCA-based toxicity analysis for the Australian urban water industry. Firstly, a review of how the gap might be bridged between LCA toxicity models and the recent knowledge generated on wastewater toxicity from bioanalytical techniques. Secondly, available LCA-based research on the environmental behaviour of metals should be included in any toxicity models that are employed. Thirdly, consideration should be given to incorporating the available scientific knowledge on risk and hazard assessment of biosolids and fertiliser metals when applied to Australian agricultural soils.

  The above focus on water system process flows does not necessarily imply they are the biggest uncertainties in the analytical framework for estimating life-cycle toxic impacts. The fourth required improvement is to the estimation of background (supply chain) toxicity. Emissions to air are the major contributor in that regard, however conventional LCA approaches rely on globalised or European fate-exposure models that appear fundamentally unsuited to Australia's unusual geographic circumstances. To the extent possible, the available models should be customised to better match these local conditions.

- Wherever possible, characterisation factors for phosphate rock depletion should be developed for contemporary metrics that assess the significance of minerals resource depletion in LCA. That will provide a starting point to consider whether LCA impact assessment models can, in fact, adequately represent the somewhat unique socio-economic characteristics of the phosphorus resource sustainability challenge.

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APPENDIX A1 STRATOSPHERIC OZONE DEPLETION & LCA

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Stratospheric ozone depletion

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Abstract
The stratospheric ozone layer plays a critical role in regulating conditions on Earth, but has been substantially depleted by CFC (chlorofluorocarbon) and other halocarbon emissions. This has increased transmission of UVB radiation to the surface, and been implicated in a range of negative human and ecosystem health impacts.

Midpoint-level LCA has traditionally utilised the steady-state Ozone Depletion Potential factors that are prominent in policy making. Current ozone-depletion endpoint models incorporate skin cancer, cataract damages, and certain changes in ecosystem productivity caused by excess UVB exposure. Other health, ecosystem and agri-production impacts are still to be incorporated into the LCA framework.

As the ozone layer recovers following regulated halocarbon emission reductions, scientific attention turns to the question of longer term ozone layer management. While growing anthropogenic emissions of N_2O (nitrous oxide) might pose a threat to ozone layer recovery, the mitigating effects of CH_4 (methane) and CO_2 (carbon dioxide) emissions will more than compensate for this. Global stratospheric ozone is expected to exceed pre-industrial levels sometime this century, albeit with a very different spatial distribution. Predictions are that UVB

levels will remain elevated in the tropics, but become depressed in other regions. That latter situation might increase the incidence of diseases associated with insufficient UVB exposure.

Whatever the policy response to these new challenges, it seems the interface of ozone layer science and management will become increasingly complex. It may be that the metrics used for ozone layer analysis will also need to evolve, if LCA is to remain relevant to this new management paradigm.

Keywords
ozone layer; cancer; chlorofluorocarbons (CFCs); carbon dioxide (CO_2); nitrous oxide (N_2O); methane (CH_4); greenhouse gases; LCA; life cycle assessment; LCIA; life cycle impact assessment

1 Stratospheric ozone chemistry

Ozone (O_3) is a natural constituent of the Earth's atmosphere, most heavily concentrated in the lower part of the stratosphere (Figure 4.1). While varying in thickness and altitude, this band of elevated ozone concentration extends around the entire globe, and is commonly referred to as the *ozone layer*.

Ozone is an extremely reactive substance, and its presence in the stratosphere is the result of a continual cycle of formation and breakdown processes. Ozone breakdown occurs both chemically and by photodissociation. This latter pathway is the only part of the ozone cycle that absorbs wavelengths spanning the UVB spectrum (Rowland 2006), intercepting the vast majority of the UVB radiation reaching the outer atmosphere. Since ~90% of atmospheric ozone resides in the stratosphere, the amount of UVB radiation reaching the Earth's surface is very sensitive to stratospheric ozone concentrations (Fahey and Hegglin 2011).

Net stratospheric ozone concentrations are strongly influenced by a small group of reaction pathways, predominantly associated with halogen, NO_x, and HO_x free radicals. These reactions are catalytic in nature, whereby the reaction step that destroys the ozone molecule is followed by another that regenerates the original free radical molecule. The stable nature of the stratosphere provides relatively little opportunity for these catalytic reaction chains to be interrupted.

The extent of degradation via these catalytic pathways is buffered by the presence of stratospheric 'reservoirs' of compounds that are unreactive with ozone ($ClONO_2$, HCl and HNO_3). The reservoir compounds bind up molecules that would otherwise be present in the form of reactive radicals (Cl^-, NO_x, HO_x), although they do break down slowly over time. The greater the relative abundance of the reservoir compounds, the lesser opportunity there is for ozone breakdown to occur (Rowland 2006).

Another controlling factor for the rate of ozone degradation is stratospheric temperature, which has a strong influence on the abundance of ozone-destroying NO radicals; and on the chemical breakdown of ozone molecules to form stable O_2.Outside of the polar regions, the influence of lower temperatures will be a net increase in ozone abundance (Bekki et al. 2011, Portmann et al. 2012).

Somewhat different processes occur in the polar regions, during the period from late winter to early spring. Conditions at this time encourage a different set of catalytic reaction chains, which are far more destructive than occur elsewhere in the stratosphere. At the same time, low UV levels limit the generation of ozone by photodissociation of O_2, and local ozone levels become severely depleted. These transient 'ozone holes' have occurred regularly over the Antarctic

since the early 1980s (Rowland 2006, Fahey and Hegglin 2011), and more recently over the Arctic region (Manney et al. 2011).

2 Anthropogenic emissions

A strong scientific consensus exists that anthropogenic emissions caused substantial levels of stratospheric ozone depletion over the latter parts of the twentieth century (WMO 2007, Douglass et al. 2011, Fahey and Hegglin 2011, Montzka et al. 2011) (see Figure 4.2).

Historically, both the scientific and policy communities have reserved the term "ozone depleting substances (ODS)" exclusively for describing the halocarbon substances that are controlled under the Montreal Protocol (see below). So as to avoid confusion with other literature, this chapter uses the more generic term of "ozone affecting substances (OAS)" - which encompasses all (halogenated and non-halogenated) emissions that can cause substantial decreases or increases in stratospheric ozone abundance. OAS emission types are discussed here in three groups.

2.1 Halocarbon emissions

While the different halogenated emission types vary substantially in their characteristics, the general pathways to ozone destruction are the same. Source gases containing chlorine or bromine are sufficiently stable to reach the stratosphere in their original form, where they break down when exposed to the high intensity of UV radiation present in the upper stratosphere. This releases the chlorine and bromine atoms, providing a source of reactive radicals that can initiate catalytic ozone destruction pathways. Halogenated substances containing fluorine or iodine molecules, but no chlorine or bromine, pose little threat to the integrity of the stratospheric ozone layer (Rowland 2006, Fahey and Hegglin 2011).

By 1987, concerns over risk to the ozone layer led to the initiation of the Montreal Protocol. This regulated the phase-out of harmful halocarbon emissions from most anthropogenic sources, notably the use of CFCs, HCFCs and Halons as refrigerants, solvents and fire extinguisher agents. These international actions halted the rapid increases in stratospheric chlorine and bromine, and global ozone levels started to recover again from the early 1990s (Figure 4.2).

Ozone Depletion Potentials (ODP) factors are ratios of the ozone change induced by a unit emission of a particular substance, benchmarked against the ozone change induced by a unit emission of CFC-11 (Solomon et al. 1992). ODP values are most commonly generated using steady state calculation approaches. This accounts for the full extent of ozone destruction caused by an emission, regardless of how long it might take for that to eventuate, and regardless of where it happens in the stratosphere. Steady state ODPs can be calculated from modelled predictions of ozone loss; or on a semi-empirical basis directly from observed data on the atmospheric behaviour of chlorine and bromine.
EESC (Equivalent Effective Stratospheric Chlorine) provides a measure of the stratospheric abundance of reactive halogens at any point in time (Newman et al. 2007). N_2O has also been successfully integrated into EESC calculations (Daniel et al. 2010). EESC is frequently used in atmospheric modelling studies for trending the stratospheric response to emission changes, and integrating these trends over timeless results are frequently expressed as global-mean values, and in this regard, are analogous to steady-state ODP calculations. For both metrics, global-mean values are weighted by surface area,

and therefore more strongly influenced by changes in mid-latitude (and tropical) regions than changes in polar regions.
Erythemal-UV is a term commonly used to express the magnitude of biologically damaging UV radiation. For this, the amount of solar radiation in different wavelengths is weighted by the potency of the UV in those wavelengths to cause sunburn (erythema). The overall intensity of erythemal-UV irradiance is most strongly influenced by radiation levels in the UVB spectrum.

Box 4.1 Commonly used metrics for analysis of ozone-affecting substances

A collection of recent studies demonstrate how important those changes were to preserving the ozone layer function. Their findings indicate the ozone layer would have collapsed if the Montreal Protocol had not been implemented (Newman et al. 2009, Garcia et al. 2012), leading to substantial increases in surface levels of erythemal-UV radiation (Newman and McKenzie 2011), and substantial increases in incidence of skin cancer (Newman and McKenzie 2011, van Dijk et al. 2013).

Even with these successes, there remains a legacy of halocarbon emissions that will continue for many years (Table 4.1). Large 'banks' of CFC and Halon substances remain in the economy, expected to leak slowly into the atmosphere over the coming decades. While less potent, the production of HCFCs is still growing in developing countries, and substantial 'banks' are being accumulated. Carbon Tetrachloride and Methyl Bromide present a somewhat different management challenge, given concerns that there are substantial sources falling outside the current regulatory regime (Yvon-Lewis et al. 2009, Xiao et al. 2010, Montzka et al. 2011).

2.2 Nitrous oxide emissions

Similar to many halocarbon OAS, nitrous oxide (N_2O) is a long-lived substance that is very stable in the troposphere. The vast majority of surface emissions will reach the stratosphere, where 90% of that N_2O will convert to stable N_2.A portion of the remainder breaks down into NO radicals that can initiate catalytic ozone destruction cycles (Rowland 2006; Portmann et al. 2012, Revell et al. 2012b).

This potential for N_2O emissions to damage the ozone layer has been recognised since the early 1970s (Crutzen 1970, Johnston 1971). Despite that, N_2O was never included in the list of controlled substances under international regulations. However, now that halocarbon emission controls have been implemented so successfully, scientific attention has returned to the role that anthropogenic N_2O emissions might play in determining the future status of the ozone layer (e.g. Ravishankara et al. 2009). Anthropogenic sources contribute approximately 40% of the current global N_2O emissions budget (IPCC 2007, Syakila and Kroeze 2011), and there is considerable uncertainty surrounding expected future growth in emission rates. Based on mid-range projections, anthropogenic N_2O emissions will exert far greater potential to deplete the ozone layer than will the remaining halocarbon emission sources (Table 4.2).

In recognition that anthropogenic N_2O emissions could impede future ozone layer recovery, the most recent synthesis from the WMO Scientific Assessment Panel has for the first time included substantive coverage of the effect that N_2O can have on the ozone layer (WMO 2011).

2.3 Carbon dioxide and methane emissions

All greenhouse gases absorb and re-emit radiation towards the Earth's surface, increasing the heat retained in the lower atmosphere (see Chapter 3 on Climate Change). The particular radiative properties of CO_2 and water vapour mean that they will also reduce temperatures in the stratosphere (Forster et al. 2011). As noted above, reduced temperatures will slow the rate of ozone breakdown in the stratosphere, leading to increased overall ozone abundance.

Atmospheric modelling studies consistently indicate that ongoing growth in CO_2 emissions will expedite ozone layer recovery; then deliver a "super recovery" (see Figure 4.2) whereby global ozone abundance goes over and above pre-industrial levels (Austin et al. 2010, Eyring et al. 2010, Oman et al. 2010, Plummer et al. 2010, Fleming et al. 2011). By the year 2100, atmospheric CO_2 is expected to be the strongest anthropogenic influence on ozone layer status (Fleming et al. 2011).

The net effect of future anthropogenic CH_4 emissions will also expedite ozone layer recovery, albeit through a complex mix of counteracting influences (Plummer et al. 2010, Fleming et al. 2011, Revell et al. 2012b). Some portion of surface CH_4 emissions will react to form water vapour, and therefore contribute to lowering stratospheric temperatures.CH_4 emissions will also have direct chemical effects (both generating and depleting stratospheric ozone by different pathways), and indirect chemical effects (e.g. impeding ozone destruction by binding reactive chlorine into HCl reservoirs)(Rowland 2006, Revell et al. 2012b). Over the longer term, ozone increases induced by anthropogenic CH_4 might approximately offset the depletion induced by anthropogenic N_2O (Fleming et al. 2011).

3 Effects on human health and ecosystems

Depletion of the ozone layer will increase the transmission of UVB to the Earth's surface, which can then cause a variety of human health and ecosystem effects (Figure 4.3).

Excessive exposure to UVB is strongly linked to risk of skin cancer and certain types of eye diseases (Norval et al. 2011, UNEP EEAP 2012), accounting for two thirds of the UV-related health burden quantified in a previous WHO study (Lucas et al. 2008). Skin cancers are the most common form of cancer in high-risk countries such as Australia, and the incidence of melanoma and non-melanoma cancers has grown in many countries over the last 4 decades (Norval et al. 2011).

A number of other health outcomes are also linked to excess UVB exposure. The incidence of Merkel Cell Carcinoma, a particularly aggressive form of skin cancer, is growing rapidly in a number of populations (Agelli et al. 2010, Girschik et al. 2011, Kuwamoto 2011, UNEP EEAP 2012). At the other extreme, relatively mild but prevalent sunburn cases could even be considered to have a substantial overall health burden (Lucas et al. 2008). Excessive UVB exposure likely contributes to other diseases of the eye, in particular pterygium and (to a lesser extent) ocular melanoma. A third category of UVB related health effects concerns the suppression of immune response to viral and bacterial infections. While the mechanisms for this effect pathway seem well understood, there is little data available to quantify the scale of this health burden (Norval et al. 2011).

Insufficient UVB exposure is also considered a public health concern in many regions of the world, particularly in winter months in the mid-high latitude zones of the globe. UVB exposure is the main source of vitamin D for many population groups, hence low exposure to UVB can induce a Vitamin D deficiency. This has been associated with a range of negative health effects, including multiple sclerosis, diabetes, and some infectious diseases (Norval et al. 2011, UNEP

EEAP 2012). While quantitative cause-effect relationships have not yet been established across this spectrum of issues, preliminary analysis illustrates that UVB deficiency could be as substantial as the human health burden that is currently associated with UVB excess (Lucas et al. 2008).

Negative implications of exposure to UVB radiation have been established for many different aquatic species at different trophic levels. However, as for terrestrial environments, there is little information available on broader ecosystem responses to UV change (Hader et al. 2011, UNEP EEAP 2012).

There have been strong causalities established between changes in UVB levels and changes in terrestrial ecosystems. Some field studies show large decreases in plant productivity, as a result of extreme changes in surface UVB radiation in high latitude regions. Others suggest that an increase in UVB levels can have positive productivity effects, by impeding herbivorous insect activity. In terms of more general ecosystem response, there is insufficient data available to establish the net scale or direction of ecological impacts associated with changing levels of UVB radiation (Ballare et al. 2011, UNEP EEAP 2012).

The effect that changes in UVB radiation will have on agri-production systems is also somewhat unclear. While there are many studies showing crop yields being impeded by increased exposure, there are also concerns that this generalised conclusion has been biased by studies of extreme and unrealistic changes (Kakani et al. 2003, Wargent and Jordan 2013). Recent analysis suggests a complex mix of positive and negative effects (Kataria et al. 2013, Martinez-Luscher et al. 2013, Mazza et al. 2013), with hopes growing that there may be opportunities to use high UVB radiation levels to enhance agri-production outputs (Wargent and Jordan 2013).

4 Spatial and temporal variability

4.1 Spatial variation

Due to the nature of circulation patterns in the atmosphere, the source location of most OAS emissions has little influence on the scale or distribution of ozone layer effects. The exception to this is a group of very-short lived halocarbons, whose chances of reaching the stratosphere will depend on the emission location and tropospheric conditions. It is thought that large quantities of brominated very-short lived substances do reach the stratosphere, although these are predominantly of natural origin (Montzka et al. 2011).

The extent of ozone layer depletion varies substantially at different latitudes (Figure 4.4), being strongly influenced by stratospheric conditions (e.g. solar radiation levels; temperature) and overall atmospheric circulation patterns.Net ozone generation rates are highest in the tropics, because of the greater temperatures and levels of solar radiation that reach the stratosphere. Ozone abundance in tropical regions has changed little over the past decades. Net depletion rates are most severe in the low temperature polar regions, particularly associated with the transient Antarctic ozone layer ‘hole’ (Fahey and Hegglin 2011). However, it is the smaller decreases at mid-latitudes that make the greatest contribution to reductions in the overall global ozone levels (see Box4.1).

A similar response can be seen in historical changes to surface levels of erythemal-UV irradiance, with the greatest increases occurring in the southern hemisphere, particularly over the Antarctic (Figure 4.5).

Demographic factors will also have a strong influence on the extent and distribution of human health effects caused by changes in UVB radiation levels. Skin colour is a major determinant for skin cancer development, with fair skinned populations showing the highest incidence rates (Norval et al. 2011). There is, however, some recent evidence suggesting that non-melanoma cancer types are more common in dark skinned populations than previously thought (UNEP EEAP 2012). Incidence of skin cancer and UV-related eye disease also increases with population age (Norval et al. 2011), suggesting that the future influence of aging populations is likely to be non-trivial (van Dijk et al. 2013). Differing cultural attitudes to sun protection might also be expected to influence the distribution of human health effects (e.g. Callister et al. 2011, Cancer Council Australia 2007, Nowson et al. 2012).

4.2 Evolving conditions over time

Most ozone affecting substances have long atmospheric lifetimes, and their effect on the ozone layer is manifested over many years. Furthermore, the extents to which halocarbon and non-halocarbon emissions will affect the ozone layer are strongly interlinked, with these interactions also playing out over long timeframes.[5.1] The future evolution of emission rates will therefore have important implications for estimating the scale of ozone layer depletion that occurs over time.

The significance of these changes can be seen in the steady-state ODP factors provided in Table 4.2. As described in previous sections, the changes in background conditions over this timespan will affect the marginal ozone destructiveness of both CFC-11 and N_2O emissions. However, the varying ODP values calculated for N_2O indicate that the two substances respond very differently to the evolving atmospheric conditions.

The changes shown for N_2O also illustrate the complex nature of the interdependencies involved. The ODP for N_2O calculated with year 2000 conditions is much lower than for earlier times, largely because the increased year 2000 levels of atmospheric chlorine, CH_4 and CO_2 all have the effect of inhibiting NO_X–driven degradation of ozone. However the situation changes looking forward to the year 2100, as these strong influences largely cancel each other out. On the one hand, further increases in atmospheric CO_2 (via stratospheric cooling) and CH_4 (via chemical effects and stratospheric cooling) will dampen the ozone potency of N_2O. On the other hand, lesser atmospheric chlorine means an increase in N_2O potency, as less reactive NO_X gets bound into $ClONO_2$ reservoirs (see Ravishankara et al. 2009, Plummer et al. 2010, Fleming et al. 2011, Portmann et al. 2012, Revell et al. 2012a).

This evolving chemistry is also expected to change the spatial distribution of ozone abundance in the stratosphere. Each of the non-halocarbon species (N_2O, CH_4, CO_2) influence the ozone layer via different pathways, and each has differing spatial implications for ozone status. Also important is the effect that anthropogenic climate change will have on atmospheric circulation patterns, increasing the rate of bulk air and ozone transfer from the tropics (where ozone generation rates are highest) towards the mid-latitudes and poles (see Portmann and Solomon 2007, Plummer et al. 2010, Bekki et al. 2011, Fleming et al. 2011, Revell et al. 2012b, Garny et al. 2013).

This complex mix of influences will change the spatial distribution of global stratospheric ozone in two important ways. Firstly, tropical ozone levels are expected to decrease below those of pre-industrial times, despite overall global ozone levels recovering strongly. Secondly,

[5.1] See Portmann et al. (2012) for a concise summary of some important interdependencies that lead to non-linear ozone responses in atmospheric modelling

ozone layer recovery will be much more rapid in the mid-latitudes than in polar regions; and occur sooner in the northern hemisphere than in the south (Austin et al. 2010, Oman et al. 2010, Plummer et al. 2010, Fleming et al. 2011, Garny et al. 2013).

Trends in surface UVB levels are expected to follow a similar (but inverse) pattern (Figure 4.5), with future UVB levels being higher in regions where they are already high (i.e. the tropics), and lower in the higher latitude regions where winter UVB levels are already low (Bais et al. 2011, McKenzie et al. 2011). This suggests the future might involve a more complex mix of human health risks related to ozone layer status - increased problems associated with UVB excess in the tropics; along with increased problems caused by UVB deficiency in other regions.

5 Midpoint assessment methodologies for LCA

Ozone Depletion Potential (ODP) factors (see Box4.1) for halocarbon emissions have been a cornerstone of midpoint-level impact assessment since the early days of the LCA methodology. The majority of LCIA methods have favoured the use of steady state Ozone Depletion Potential (ODP) values, adopting the halocarbon factors that are updated periodically by the WMO.

It has been rare for non-steady state ODP values to be actively promoted for consideration in midpoint analysis, although the EDIP97 method (Hauschild and Wenzel 1998) and the CML-IA method (Guinée et al. 2002)do include a set of timeframe-specific factors published in the early 1990s (Solomon and Albritton 1992). Unfortunately, there has been little further development of non-steady state approaches in the published literature since that time. As a consequence, LCA practitioners do not have ready access to timeframe-dependent ODPs that also incorporate the latest knowledge on atmospheric behaviour of the different ozone-affecting substances.

There is a generally high degree of confidence ascribed to the ozone layer predictions that are synthesised through the WMO Scientific Assessment Panel review process (Daniel et al. 2011), even though certain fundamental uncertainties in the available atmospheric models are acknowledged (e.g. Bekki et al. 2011; Thompson et al. 2012). Much of these uncertainties are thought to affect the modelling of different halocarbons in a similar manner, and therefore cancel out when the ODP factors are calculated relative to a reference halocarbon substance.

The use of steady-state ODP factors for midpoint LCA modelling continues to be a highly policy-relevant approach. ODP values are formally adopted under the Montreal Protocol, and regularly used for national and international reporting on aspects to do with ozone layer management. In the most recent review from the WMO Scientific Assessment Panel, the value of the ODP metric was again promoted because of its simplicity and transparency (Daniel et al. 2011). That report contains the most up to date set of steady state ODP factors for halocarbon emissions, reflecting the latest developments in atmospheric modelling capability.

However, it remains to be seen whether the conventional ODP metric will retain its relevance into the future. The focus on halocarbon emissions in the WMO assessment reports is in part a legacy of the Montreal Protocol mandate, and does not necessarily imply that halocarbon emissions should still be the highest priority concern. Attention in the science community is now shifting more towards the longer term management of ozone layer recovery, and it is less clear how useful the ODP metric will be in this context.

6 Endpoint assessment methodologies for LCA

Early attempts by the LCA research community to model the damages related to stratospheric ozone depletion (e.g. Steen 1999; Goedkoop and Spriensma 2000) were hampered by a lack of quantitative science in formats that could readily support the development of LCA impact models.

More recently, two new approaches to LCA damage assessment have incorporated substantial improvements in basic understanding of the process and effects of ozone layer depletion.

The first of these was the LIME method (Hayashi et al. 2006), which provides characterisation factors for halocarbon compounds. The LIME model uses regressions to link emissions to changes in global ozone, then to changes in surface levels of UVB radiation. Human health damages (in DALYs) are calculated by combining distributions for this change in UVB radiation, with global distributions of skin colour (for skin cancer) and age (for cataracts), with literature based dose-response functions.

A strength of the LIME method is that it provides great breadth in the range of effects that are covered. Characterisation factors are also included for the impacts of UVB changes on net primary productivity for terrestrial (lowland conifer forest) and high-latitude aquatic (phytoplankton) ecosystems. The LIME method also characterises impacts on 'social assets', providing results for UVB-induced changes to yields of global food crops and managed timber production.

The second endpoint model of note is from Struijs et al. (2010)[5.2].That study also focused exclusively on halocarbon substances, but differs from the LIME method in that it provides characterisation factors only for the human health effects of skin cancer and cataracts.

The Struijs et al. (2010) modelling of skin cancer and cataract effects is based on a much higher spatial resolution for estimating the degree of human exposure to increased UVB radiation, and a more sophisticated treatment of demographics. They also make use of global scale models (van Dijk et al. 2008) to predict the evolution of surface erythemal-UV distributions, accounting for the latest available empirical data as well as climate-sensitive predictions of future cloud cover.

The Struijs et al. (2010) model also has the advantage of being well-aligned with best practice approaches to estimating changes in ozone abundance. Fate factors are calculated from time-integrated estimates of change in global-mean EESC (see Box4.1). This allows a more robust consideration of the ozone response for longer lived substances, whose destructiveness might change over time as background stratospheric conditions change. The EESC modelling approach adopted standard conventions to minimise uncertainty, although there have been changes to the recommended best-practice modelling parameterisation (see Daniel et al. 2011) since those EESC simulations were undertaken.

More fundamental challenges are introduced by the need to calculate absolute, rather than relative, measures of ozone change. In this regard, the strong influence of stratospheric temperature and atmospheric circulation patterns represent notable sources of uncertainty. The available atmospheric models do provide fairly consistent conclusions in regard to both these issues. However, calibration of the models against empirical temperature and circulation rate

[5.2] The Struijs et al. (2010) model evolved from the version provided with the ReCiPe method (Struijs et al. 2009). Only the more recent version is discussed here - as it contains a number of notable improvements, and the two models are otherwise structurally very similar.

data has been problematic, and this remains a substantial concern (Bekki et al. 2011, Thompson et al. 2012).

Uncertainties are also introduced when translating ozone change into quantitative predictions for disease-relevant levels of UVB radiation. This step is made difficult by limitations in the ability to model the influence of other confounding factors (e.g. cloud cover), and uncertainty over the spectrum of wavelengths most relevant to each of the different endpoint effects (McKenzie et al. 2011). Additional complications exist for estimating exposure to the eyes, where quantified surface levels of UVB radiation are not necessarily a good proxy for exposure (McKenzie et al. 2011, Norval et al. 2011). Furthermore, the influence of changing social behavioural patterns is likely to be strong (Norval et al. 2011).

For the damage pathways that are included in the two 'best-practice' endpoint LCA models (skin cancer; cataracts; terrestrial productivity response; aquatic productivity response), there has been a considerable increase in the body of published data since they were developed. Notwithstanding that, many important data gaps remain. Even for skin cancer, the most well studied of this group, the lack of data from developing countries and dark-skinned populations is seen as a concern (Lucas et al. 2006, Wright et al. 2012).

Other uncertainties in the Struijs et al. (2010) calculation of ozone, and changes in UVB exposure, are introduced with choices made in their modelling design. These are discussed further in Section7.

7 New developments and research needs

7.1 Including non-halocarbons in LCA analysis of stratospheric ozone depletion

Conventional LCA already has a strong focus on emissions of N_2O, CH_4 and CO_2, due to their links to climate change. Also including these substances in life-cycle ozone depletion assessment could potentially change the conclusions that might be drawn in certain studies (e.g. Lane and Lant 2012). Doing so would be methodologically consistent with the marginal-assessment approach favoured in Life-cycle Impact Assessment (see Huijbregts et al. 2011), given these substances will be the key drivers of ozone layer status into the future.

While there is not yet any policy imperative for such a change, it remains to be seen how the international community will respond to the growing scientific focus on the importance of non-halocarbon emissions. For N_2O, there have already been calls for the Montreal Protocol to be used to regulate anthropogenic emissions (Kanter et al. 2013).

If N_2O, CH_4 and CO_2 are to be included in conventional LCIA of ozone layer effects, then further critical review is required into the appropriate means for doing soothed atmospheric modelling community has been cautious in their recommendation of the available ODP factors for non-halocarbons, because the ozone-layer effects of N_2O, CH_4 and CO_2 are manifested in very different ways to those of halocarbons. This disparity increases the chance that substance comparisons could be unevenly affected by inherent bias in the underpinning atmospheric models (Daniel et al. 2011, Fleming et al. 2011).

Additional complications exist when considering CO_2 and CH_4 emissions. Negative ODP factors have been calculated for these two substances, reflecting their contribution to increasing overall global ozone levels (Fleming et al. 2011). While not a common practice, it is perfectly feasible for negative characterisation factors to be used in LCA(e.g. De Schryver et al. 2009).

Conventionally calculated ODP values might, however, have limited relevance as a proxy for the effects of CO_2 and CH_4 emissions. These substances will be the primary driver of any future ozone layer 'super-recovery', which in turn will have very different human health implications (diseases associated with insufficient exposure to UVB radiation) than those associated with no-recovery (diseases associated with excessive exposure to UVB radiation). Combining both these impact pathways might require a holistic reconsideration of the design of LCA metrics (midpoint and endpoint) for assessing ozone-affecting substances.

7.2 Sensitivity to changing atmospheric conditions

Forecast atmospheric changes over the course of the twenty-first century will substantially change the conditions that regulate the stratospheric ozone cycle. For all halocarbon and non-halocarbon OAS, this will mean changes in the extent to which current and future anthropogenic emissions impact on the ozone layer. Furthermore, the different substances will respond differently to these changing stratospheric conditions. Because the substance interdependencies are so strong, predicting the ozone depletion caused by a marginal OAS emission will also be very dependent on the assumptions made about future emission trends for halocarbons, N_2O, CH_4 and CO_2.These emission forecasts are themselves a source of great uncertainty.

Steady state ODP factors are calculated on the basis that atmospheric conditions remain constant over time, and are therefore unable to capture such complexity. CA practitioners may therefore benefit from having access to a number of possible characterisation factor sets, each reflecting a different set of underpinning atmospheric conditions.

This would not, however, address the fundamental discrepancy between the steady-state calculation approach and the fate factors used in the Struijs et al. (2010) endpoint model. The latter were modelled for an evolving set of atmospheric conditions in response to a temporally varying profile of halocarbon and other greenhouse gas emissions. These midpoint and endpoint approaches could potentially reach very different conclusions when comparing substances that respond differently to changing background conditions, whether that is because of different atmospheric lifespans (e.g. CFC-12 vs. HCFC-123), or because they are affected by different chemical pathways (e.g. CFC-12 vs. N_2O).

7.3 Including 'post-recovery' effects

Another source of inconsistency between midpoint and endpoint metrics, once again related to differing approaches to time-integration, would be the choice to truncate the life cycle impact assessment at some pre-conceived point of ozone layer 'recovery'.

The defining feature of steady-state ODP factors is that they account for the full amount of ozone destruction caused by an emission, regardless of whether that is expressed over 1 year (e.g. HCFC-123) or over 100 years (e.g. CFC-12). In contrast, the Struijs et al. (2010) endpoint model truncates damage assessment at the year 2049, on the premise that health effects beyond that date would be at 'normal' background levels. Using that approach, endpoint analysis would only account for some portion of the ozone depletion impacts caused by longer lived emissions, but all of those caused by shorter-lived emissions. Once again, it seems likely that these midpoint and endpoint approaches could reach different conclusions when applied in certain contexts.

A further complication with the truncation approach is that the choice of 'recovery' benchmark is not straightforward. Three issues warrant consideration in this regard. Firstly, the policy

convention of benchmarking recovery against conditions in the year 1980 (as followed by Struijs et al. 2010) represents a somewhat arbitrary choice for LCA. Many recent studies indicate that ozone levels in 1980 were already impacted by anthropogenic activity (Austin et al. 2010, Eyring et al. 2010, Oman et al. 2010, Plummer et al. 2010, Fleming et al. 2011, Portmann et al. 2012). Secondly, the timing for a return to pre-industrial levels of global ozone will be strongly influenced by the choice of greenhouse gas emission scenarios (Eyring et al. 2010). Thirdly, this benchmarking of global ozone abundance disguises the expectation that 'recovery' dates will vary substantially at different latitudes. Recovery over the northern hemisphere mid-latitudes, where much of the world's fair skinned population resides, will happen much sooner than elsewhere on the globe. At the other extreme, ozone abundance over the tropics might still be lower than pre-industrial levels, well into the 22nd century.

This complexity suggests that LCA practice would benefit from having a choice of characterisation factors, reflecting (a) different timeframes of concern, and (b) different forecasts for the evolution of greenhouse gas emission rates. It also raises the question of what to do about the likelihood of 'post-recovery' health effects associated with low (insufficient) exposure to UVB radiation.

7.4 Using time-integrated midpoint characterisation factors

As an alternative to using steady-state calculations, midpoint level characterisation factors could instead be generated with time-integrated modelling applied over defined timeframes. The latter approach is already the norm for midpoint-level analysis of climate change impacts. Timeframe-dependent ODP calculations are methodologically possible (see Solomon and Albritton 1992), and could potentially provide some flexibility in how the issues of evolving background atmospheric conditions (Section7.2) and recovery date truncation (Section7.3) are considered.

Given the likely sensitivity of this approach to the choice of greenhouse gas emission scenarios, consideration should be given to maintaining consistency with any assumptions used for developing climate change characterisation factors.

7.5 Spatially resolved modelling of changes in UVB levels

The two most current endpoint-level LCA models both estimate the change in harmful UV levels based on historical relationships between global ozone abundance (or its proxies), and the spatial distribution of UVB radiation levels at the surface. Both assume that the spatial distribution of marginal changes in these UVB radiation levels will be the same for all emission types.

This approach does not reflect the expectation that future surface UVB distributions will evolve in a very different manner to the way they changed in the past. Furthermore, the assumption that all substances have a similar spatial influence may be less valid if non-halocarbons are to be considered in the analysis of ozone depletion effects.

Further investigation would be required to identify whether or not these differences would have a material effect on calculated endpoint characterisation factors; and how sensitive this might be to changing greenhouse gas emission scenarios.

7.6 Estimating the damages

Opportunities exist to improve on the dose-response information used to convert exposure rates into estimates of health damage, and to extend the range of disease types that are considered in LCA. Recent years have delivered substantial new insight into trends of melanoma and non-melanoma skin cancer incidence, and also for the growing occurrence of Merkel Cell Carcinoma. Previous quantitative studies (e.g. Lucas et al. 2008) might provide a template for LCA endpoint analysis to include less severe effects such as sunburn, pterygium, and recurring viral infections.

Endpoint damage modelling will also need to adequately consider the substantial uncertainty involved in predicting future behavioural patterns. Behavioural change could potentially have a very significant influence on the human health burdens associated with UVB exposure.

For the consideration of ecosystem and agri-system effects, existing endpoint LCA models should be updated to better reflect contemporary thinking. Many new studies exist to inform the quantification of plant response to changes in UVB radiation levels, although the weight of conflicting evidence may make it difficult to produce generalised dose-response functions.

Finally, for LCA endpoint analysis to embrace a new phase of ozone layer management, a more fundamental reconsideration of endpoint modelling priorities might be required. First of all, greater attention to human and ecosystem effects in tropical zones might be warranted, given the expectation that UVB levels in that region will remain elevated for the foreseeable future. Secondly, growing concerns about the health implications of under-exposure to UVB radiation suggest that this impact pathway should be considered for inclusion in the LCA framework.

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Prof. Robyn Lucas, of the Australian National University, provided guidance on the topic of human health damages, and useful feedback on a draft of this chapter. Her contributions were greatly appreciated. Thanks also to Prof.AlkisBais (AUTH) for assistance with producing Figure 4.5.

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Tables

Table 4.1 Forecast anthropogenic emissions of halocarbons and N_2O over the period 2011-2050; and key atmospheric characteristics of each substance type

Emission group	**Time-integrated, ODP-weighted, emission forecasts (Mt CFC-11-eq)[a], and the main anthropogenic emission sources**		**Atmospheric lifetime (y)** [b]	**Steady-state ODP (kg-CFC11e/kg)[b]**
CFC banks	1.27 Mt	Refrigerants and other compounds stored in goods that were manufactured prior to the CFC phase out	45 - 1020	0.57 – 1.0
Halon banks	1.09 Mt	Fire extinguishers manufactured prior to the Halon phase out.	16 – 65	3 – 10
HCFC production and use	0.66 Mt	Production and use as industrial solvents; Production of refrigerants and other compounds.	1.3 – 17.2	0.01 – 0.12
HCFC banks		Refrigerants and other compounds stored in goods that were manufactured prior to the CFC phase out		
Carbon Tetra-chloride (CCl_4)	0.54 Mt	Use as industrial solvent and feedstock for chemicals manufacture[c]	26	0.82
Methyl Bromide (CH_3Br)	0.26 Mt	Use for Quarantine and pre-shipment services (QPS) [d]	0.8	0.66
Methyl Chloroform (CH_3CCl_3)	0.004 Mt	Production and use as industrial solvents.	5	0.16
Nitrous Oxide (N_2O)	6 Mt	Agricultural fertiliser use; Biomass burning; Fuel combustion; Industrial processes	114	0.017 - 0.019

[a] Emission forecasts adapted from Table 5-4 of Daniel et al. (2011)

[b] Lifetime estimates and semi-empirical ODPs for the halogenated compounds were taken from Table 5-1 of Daniel et al. (2011). For N_2O, the atmospheric lifetime value is taken form table 2-14 of IPCC (2007). Three steady-state ODP values for N_2O were found in the literature – all for year 2000 atmospheric conditions, albeit calculated using different atmospheric modelling packages (Ravishankara et al. 2009, Daniel et al. 2010, Fleming et al. 2011).

[c] The CCL_4 estimate makes an allowance for unattributed emissions not being accounted for using (bottom up) emission inventories consistent with the Montreal Protocol (Montzka et al. 2011).

[d] QPS uses are exempt from the Montreal Protocol. Other notable anthropogenic sources of CH_3Br are not included in this estimate – e.g. emissions from the growth of certain crops, and from the combustion of biofuels and leaded gasoline.

Table 4.2 Steady state ODP factors (kg-CFC11e/kg) compiled from Tables 1 and 2 of Fleming et al. (2011). Ozone changes were modelled for an emission pulse in the year shown, assuming the background conditions for that year remain constant.Each ODP factor was calculated relative to the modelled ozone change induced by an emission of CFC-11 in that same year.The year 2100 atmospheric conditions are based on the WMO-A1 (WMO 2007) and IPCC-A1B (IPCC 2000) scenarios

Year	1850	1950	2000	2100
CFC11	1	1	1	1
N_2O	0.025	0.024	0.019	0.018

Figures

Figure 4.1: Vertical profile of global mean ozone abundance across the troposphere and stratosphere, taken from Fahey and Hegglin (2011). The troposphere extends from the Earth's surface to an altitude of 10-15km, and contains the turbulent atmospheric activity that defines surface weather patterns. The stratosphere is a much more stable environment, extending to approximately 50km above the Earth's surface. The majority of atmospheric ozone is concentrated between the altitudes of 15 and 35km – this band is known as the ozone layer

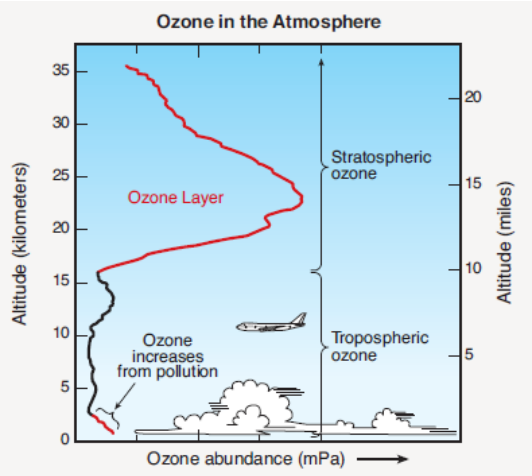


Figure 4.2 Global mean ozone estimates from NIWA satellite observations (+) and modelled time series (—) as presented in Portmann et al. (2012). These are compared against the modelled value for the year 1900, shown as the horizontal straight line. The modelled results were generated using the mid-range emissions estimates of the IPCC A1B/WMO A1 scenario (from IPCC 2000, WMO 2007)

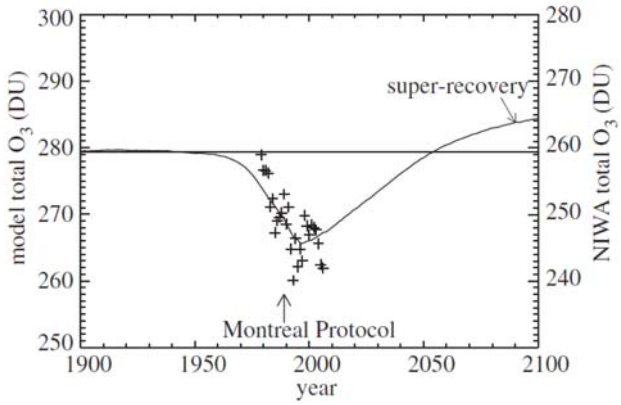


Figure 4.3 A simplified selection of impact pathways associated with anthropogenic emissions and their effect on the stratospheric ozone layer. Those pathways included in conventional LCA impact assessment models are highlighted in grey. Pathways not shown include the potential for decreased stratospheric temperature to enhance the formation of polar ozone layer 'holes'; and the climate-change mediated effect on atmospheric circulation patterns that will influence the spatial distribution of stratospheric ozone abundance

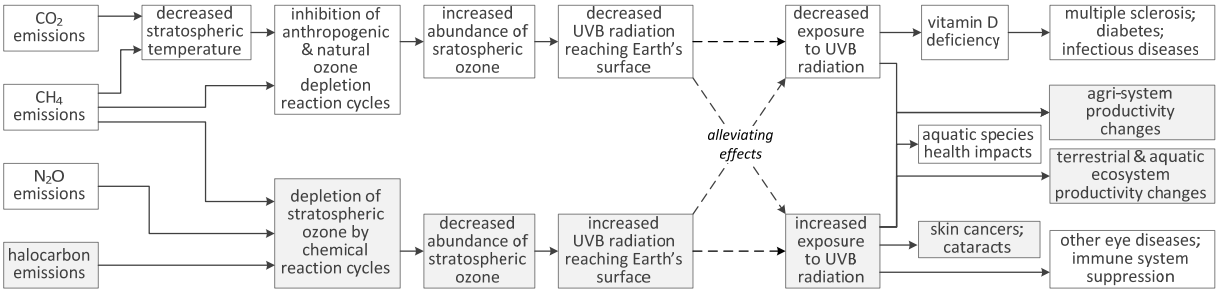


Figure 4.4 The distribution (by latitude) of ozone layer depletion, from Fahey and Hegglin (2011). These changes are based on measured, rather than modelled, datasets for annual-averaged, total column ozone levels

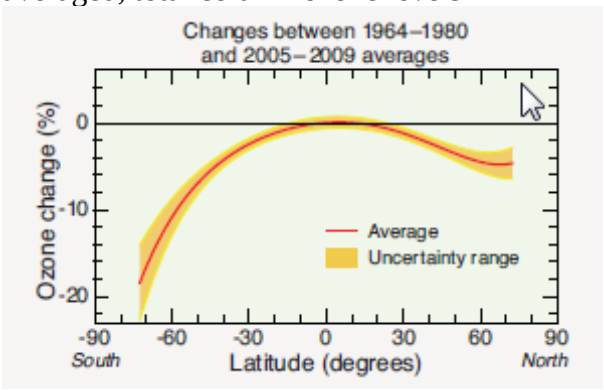
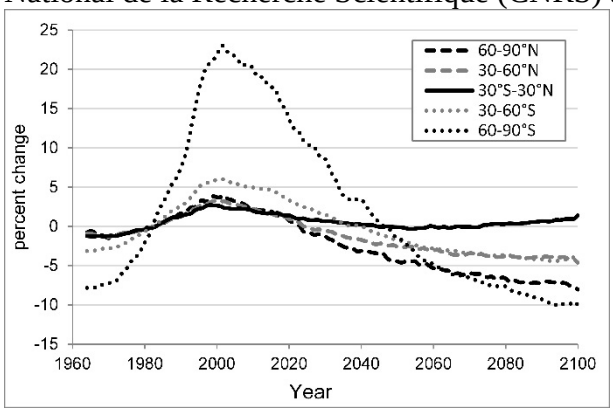


Figure 4.5 Modelled trends of surface UV for the erythemal spectrum (wavelengths relevant to skin cancer), excluding the potentially important influence of changes in cloud cover, surface reflectivity or tropospheric aerosols. Adapted from Fig.4 of McKenzie et al. (2011), with the permission of The Royal Society of Chemistry (RSC) on behalf of the Centre National de la Recherche Scientifique (CNRS) and the RSC



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APPENDIX A2 OZONE DEPLETION ANALYSIS FOR URBAN WATER SYSTEMS

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Including N_2O in ozone depletion models for LCA

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Abstract
Purpose Recent literature has highlighted a renewed debate amongst the scientific community about the relevance of nitrous oxide (N_2O) emissions to future ozone layer management. This raises the question as to whether the life cycle assessment (LCA) community should also consider incorporating N_2O into its ozone depletion models. This discussion summarises a preliminary investigation into the justification for doing so.
Methods Literature on the atmospheric science of ozone depletion and N_2O was reviewed, in particular recent proposals for an ozone depletion potential (ODP) factor that can be applied to anthropogenic N_2O emissions. To identify their potential significance to life cycle impact assessment (LCIA) results, these ODP values were applied to both a wastewater management case study and global emissions inventories. The literature review was also used to highlight certain issues that need further consideration if N_2O is to be incorporated into LCIA models.
Results Atmospheric modelling has shown that continued anthropogenic N_2O emissions could substantially affect ozone layer recovery. Furthermore, N_2O now represents one of the biggest remaining opportunities for emissions abatement. The two steady state ODP factors for N_2O available in the literature are in close agreement, with one of the models used showing reasonable calibration to accepted ODP values for other substances. Analysis of the wastewater case study showed that the incorporation of these interim ODP values for N_2O could have a substantial impact on LCIA results interpretation. This finding should be equally relevant for other case studies where N_2O emissions play a prominent role.
Conclusions The inclusion of N_2O into marginal-impact LCIA ozone depletion models would seem justified, given the relevance of N_2O emissions to a number of planning debates in which LCA currently has a prominent role. If this is not pursued, then the use of LCIA to support decision-making could mask, rather than reveal, an issue that may be environmentally relevant. Published ODP values for N_2O could be used as an interim measure. However, they are dependent on assumptions that may not be the most relevant choice for application to LCA studies. Further investigation is therefore required on how best to specify a range of ODP values for N_2O that can support robust sensitivity analysis in LCIA. Fortunately, the state of atmospheric modelling science would seem sufficiently mature to be able to inform this process. LCA-specific methodological challenges (e.g. choice of time frames, spatial implications) will also need to be addressed.

Keywords LCA · N_2O · Nitrous oxide · Normalisation · ODP · Ozone layer · Wastewater treatment

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1 Introduction

1.1 N_2O and the ozone layer

Atmospheric scientists have long recognised the damaging effect that nitrous oxide (N_2O) emissions can have on

the stratospheric ozone layer (Crutzen 1970; Johnston 1971; Kinnison et al. 1988). Although stable in the lower atmosphere and slow to break down upon reaching the stratosphere, N_2O is the main source of reactive nitrogen oxides that deplete global ozone concentrations at altitudes between 25 and 35 km. However, this risk was never formally recognised through inclusion in the list of substances targeted by the Montreal Protocol. International action to restore the ozone layer has instead focussed on chlorine- and bromine-containing compounds. These are more significant at other altitudes, plus are the dominant cause of the ozone layer hole over the polar regions (Chipperfield 2009).

Addressing N_2O emissions might seem a far less tractable problem for policy action than for the higher profile chlorinated and brominated compounds. The main concerns in the latter group are anthropogenic emissions that were, in many cases, relatively easy to target (through their association with specific technologies) and curb or eliminate. In contrast, the IPCC (2007) estimates that less than 40% of global N_2O emissions are from anthropogenic sources, with the majority of these generated from biological processes that are stimulated by nitrogen inputs to agricultural lands (~16% of total) and subsequently to waterways (~10% of total). These emissions will be highly dispersed, highly variable and difficult to estimate. Further, their association with food production means that management change is not likely to come easily or rapidly.

However, this is not a good reason to ignore the issue. Modelling by Ravishankara et al. (2009) suggests that anthropogenic N_2O emissions are currently the greatest source of human-induced stratospheric ozone layer damage. Given that the remaining stockpiles of ozone-depleting halocarbons will progressively be exhausted, and global N_2O emissions are likely to remain steady or grow, they also show that anthropogenic N_2O may become the dominant ozone depletion potential (ODP)-weighted emission in the future. Daniel et al. (2010) confirmed that, when the effects are integrated over the remainder of the 21st century, reducing anthropogenic N_2O emissions could be an effective way of expediting ozone layer recovery. It is worth noting, however, that the benefits possible from N_2O mitigation are small compared to those already delivered through ratification of the Montreal Protocol (Daniel et al. 2010). Nonetheless, there are now a number of studies (Daniel et al. 2010; Eyring et al. 2010; Plummer et al. 2010; Ravishankara et al. 2009) indicating that a continuation of anthropogenic N_2O emissions could delay and/or alter ozone layer recovery. In line with this, the most recent Scientific Assessment of Ozone Depletion by the World Meteorological Association (WMO 2011) has for the first time included substantive coverage of the implications for ozone layer protection from N_2O emissions.

1.2 The role of LCA

These recent findings may well lead to increased attention from policy makers. In the meantime, however, the historical focus on managing chlorine- and bromine-containing substances would seem to have contributed to a lack of awareness amongst the broader community on the ozone-related risks associated with N_2O emissions.

While the life cycle assessment (LCA) methodology is widely used to introduce environmental issues that would otherwise be overlooked in the decision-making process, it typically does not address this particular concern. The most common life cycle impact assessment (LCIA) midpoint-level models for ODP are reasonably consistent in their treatment of chlorine- and bromine-containing substances and use of World Meteorological Organization (WMO)-recommended ODP factors,[1] but none consider the potential for ozone layer damage caused by emissions of N_2O (ECJRC 2010). This consistency no doubt reflects the strong historical consensus on the need to mitigate emissions of chlorine- and bromine-containing substances and on the ODP metrics to inform policy responses. However, scientific comment (Chipperfield 2009; Wuebbles 2009) and literature (Daniel et al. 2010; Eyring et al. 2010; Plummer et al. 2010) following the Ravishankara study shows that debate is unresolved on the issue of N_2O emissions management.

From our perspective as users of LCA, we therefore feel that the exclusion of N_2O (from ODP models) compromises two of the fundamental benefits that LCA can offer. The first is that it provides quantitative analysis across a broad spectrum of environmental concerns, thereby providing a robust framework for including *as many environmentally relevant issues* as possible into the decision-making process. Secondly, it does this using best-estimate models of marginal impact developed with *as much scientific rigour as is possible* in each case. LCA does not provide environmental impact assessment in the strict sense, as this requires a higher level of certainty about the extent of environmental or human health damage. Rather, we see its role is to provide credible perspective on the *possible* impacts associated with a broad range of activities across the full life cycle of the products or services in question.

2 Case study application

Ravishankara et al. (2009) postulated that an ODP factor for N_2O emissions might be 0.017 kg CFC-11e/kg N_2O, much

[1] Most of the publicly available LCIA midpoint models use ODP factors taken from the 2002 WMO Scientific Assessment (Montzka et al. 2003) and have not yet incorporated the ODP changes recommended in the 2006 (Daniel et al. 2007) or 2010 (Daniel et al. 2011) scientific assessments.

lower than for most CFCs, but at a similar level to some HCFCs that are being phased out under the Montreal Protocol. Using a different model but for the same atmospheric conditions, Daniel et al. (2010) subsequently calculated a value of 0.019 kg CFC-11e/kg N_2O. To our knowledge, these are the only two literature examples of detailed atmospheric modelling being distilled into ODP values for N_2O. In doing so, they have made it possible to review the significance of this issue to LCA results. While we are not the first to identify this opportunity (see Andrae and Andersen 2011), we are not aware of any other studies that have done so in a quantitative manner. We have therefore incorporated an ODP factor for N_2O emissions into our LCIA analysis, using 0.018 kg CFC-11e/kg N_2O as the average of these two reported values.

Taking the normalisation inventories used for the ReCiPe LCIA method (Goedkoop et al. 2009; Sleeswijk et al. 2008), inclusion of this factor for N_2O would increase the ODP estimate for the overall global and European economies by 198% and 455%, respectively. Such a large change suggests that LCA results for case studies involving N_2O emissions in the foreground inventory may change substantially. Amongst others, this could affect studies on agriculture, forestry, land use change, biofuels, wastewater management and certain industrial processes.

More broadly, it could affect any studies that do (or should) include the ODP impact category and, to assist in interpreting the study results, use a normalisation step based on economy-level impact estimates. This would typically involve the case study's ODP impact score being divided by the ODP impact score for the chosen benchmark economy, so as to give some perspective on the scale of the case study result. As demonstrated above, the inclusion of N_2O will lead to large increases in the economy-level ODP estimates and therefore the denominator of this calculation. This would mean a significant decrease in the normalised score for any case study that does not itself involve significant levels of N_2O emissions. However, for those industries where N_2O emissions are likely to be substantial, the opposite effect on the normalisation process might be expected.

As an example of the latter, Fig. 1a demonstrates that including an ODP factor for N_2O can make this issue look much more significant for LCIA of an urban sewage treatment plant (STP). Here, the STP inventories are taken from Foley et al. (2010), with the midpoint impact results benchmarked against ReCiPe's estimate of the LCIA burden for the global economy. For an STP with marine discharge of treated effluent containing a total nitrogen (TN) concentration of 10 mg/L, the default ODP result ranks 16th out of 18 impact categories and is three to four orders of magnitude lower than those for Marine Eutrophication Potential and Global Warming Potential. Notwithstanding concerns about hidden bias in drawing such a conclusion (Heijungs et al. 2007), this result implies that the STP contribution to the global ozone depletion 'problem' is trivial compared to its contributions to other environmental challenges. However, if the proposed ODP factor for N_2O is included in the assessment, then the STP's contribution to global ODP ranks 4th and is on a par with those emerging issues (e.g. global warming, ecotoxicity) already of some concern to urban water system planners.

Figure 1b then illustrates how the inclusion of N_2O in ozone depletion considerations could affect LCIA-based decision-making when options are being compared. In this instance, we explore tradeoffs associated with the inexorable push to improve waterway health in Australian urban areas by reducing STP effluent nutrient levels. Environmental debate on such a decision would typically extend no further than the consideration of greenhouse gas implications. For scenarios 5 and 6 of Foley et al. (2010), reducing the TN

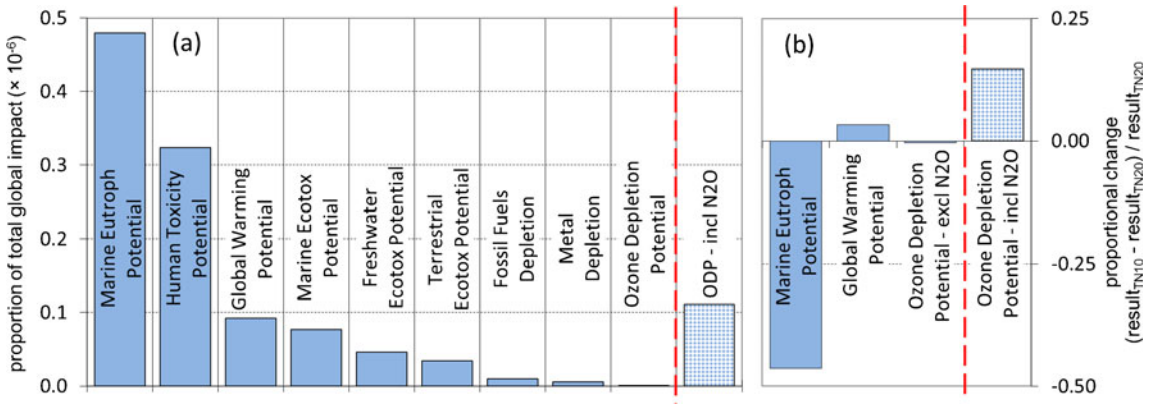

Fig. 1 Selected midpoint LCIA results applying the ReCiPe midpoint (H) method to STP scenarios 5 and 6 from Foley et al. (2010). The ODP results are compared with/without the inclusion of an ODP factor for N_2O (0.018 kg CFC11e/kg N_2O): in **a**, on an absolute basis for a scenario with total nitrogen (TN)=10 mg/L, normalised against global impact estimates, and in **b**, as a proportional change caused by a reduction in effluent TN from 20 to 10 mg/L

concentration in the effluent from 20 to 10 mg/L increases the estimated life cycle N_2O emissions by 19%. However, because of the significant GWP contribution from power use and other sources, the associated increase in Global Warming Potential is very minor compared to the reduction in Eutrophication Potential. This result would suggest only a negligible trade-off and therefore great benefits in moving to the lower effluent nitrogen concentration.

While consideration of the default ODP results would do nothing to change the conclusions drawn, inclusion of an ODP factor for N_2O may. Figure 1b shows that doing so would not only change the ranking of the two scenarios in terms of ODP but also identifies a potentially significant downside to the advanced nutrient removal. While real-life decision-making would of course be more complex than this simplified example, it does illustrate that ozone depletion results could possibly affect planning outcomes when there are substantial N_2O emissions involved.

3 Conclusions and recommendations

The presence of a notable quantitative result for ODP, whether normalised against economy-level or business-as-usual benchmarks, does not in itself confer environmental relevance. If this is a debate of little consequence because the ozone layer 'problem' has been resolved, then it would be reasonable to exclude or downplay the ODP results in most LCA case studies. However, our reading of the scientific literature is that this can no longer be considered the obvious position to take. Furthermore, the dominant paradigm in LCA is to use impact models based on estimates of marginal change (Huijbregts et al. 2011). If LCA practitioners choose (for good reason or bad) to consider midpoint ODP results, then it would seem beneficial to account for N_2O as a significant marginal risk under current circumstances.

If ODP models in LCIA are to include N_2O, then our review has highlighted five important issues that will need to be addressed.

1. If N_2O comprises a major contribution to ODP results, there will be a strong imperative for LCA practitioners to utilise high-quality N_2O emissions data. Given the large uncertainties associated with estimation from the majority of N_2O emission sources, this will not be a trivial exercise. Fortunately, research efforts in this area have accelerated in recent times thanks to growing interest from a greenhouse gas perspective. For the continental or global level inventories used in LCIA normalisation, emission estimates might more easily benefit from calibration against measured atmospheric concentration changes. Given recent developments in this regard (Corazza et al. 2011; Thompson et al. 2011), there might be some benefit in reviewing whether the available normalisation inventories for N_2O can be improved.
2. An increased focus on ODP results in LCA should also prompt a review of normalisation inventories for other ozone-depleting substances. Major distinctions can be drawn on the timing of impacts associated with those substances still of concern to policy makers. Some (e.g. HCFCs) are still in production with significant delays expected before the full emission discharges are exhausted; others already banned (e.g. CFCs and halons) will continue for some time as a source of emissions from the 'bank' of chemicals stored in products manufactured in the past, while others (e.g. CH_3Br and CCl_4) are more akin to instantaneous emissions from ongoing activities (Daniel et al. 2010, 2007). This variation in emission time lags introduces a significant methodological choice for LCIA normalisation inventories that are typically determined for a specific reference year. For example, LCIA normalisation against benchmarks based on reference year *emission* estimates (e.g. Bare et al. 2006; Lautier et al. 2010) may give very different perspectives to those based on reference year *production* estimates (e.g. Laurent et al. 2011; Sleeswijk et al. 2008).
3. Further investigation would be required to finalise the setting of N_2O characterisation factors in the midpoint ODP models used by LCA practitioners. Some confidence is provided by Ravishankara et al. (2009), whose calculated ODPs for CFC-12 (1.03) and HCFC-22 (0.06) were close to the WMO-recommended values at the time (1.00 and 0.05 respectively). Also, both the published ODP factors for N_2O were generated with steady state modelling for N_2O (Daniel et al. 2010; Ravishankara et al. 2009), consistent with the favoured approach in LCA. However, the models employed were not strictly the same as those used to generate the WMO-recommended ODP values and, by inference, the other ODP characterisation factors commonly adopted in LCIA. The relevance of their methodologies to LCA application therefore warrants closer scrutiny.
4. Furthermore, predicting the long-term potency of N_2O will depend on forecasts for a range of extenuating factors. For example, the net ODP of N_2O will be influenced by atmospheric chlorine concentrations—the values presented above (0.017 and 0.019 kg CFC-11e/kg N_2O) were both calculated for atmospheric conditions in the year 2000, at which time the stratospheric concentration of reactive chlorine was very near its historical peak (Austin et al. 2010; Clerbaux et al. 2007). But Ravishankara et al. (2009) also calculated that the ODP for N_2O would be 50% higher if stratospheric chlorine concentrations were at pre-industrial levels.

Modelling by Austin et al. (2010) suggests this may occur around the end of the century, meaning that current atmospheric conditions may not be that relevant for time-integrated LCIA modelling based on long-term (100 years or greater) time frames. A contrasting example is climate change, which will have a dampening effect on the ODP of N_2O. Plummer et al. (2010) show that, with moderate growth in greenhouse gas emissions to 2100 following the SRES A1B scenario of the IPCC (2007), the resulting atmospheric temperature changes would reduce by more than 50% the expected increase in atmospheric reactive nitrogen caused by N2O emissions.[2] The modelling of ODP characterisation factors for use in LCA should therefore be based on choices (e.g. time frames and emission scenarios) that are consistent with those underpinning the modelling of (a) ODP values for other substances and (b) characterisation factors for other LCA impact categories.

5. Finally, there is also a need to consider the implications for endpoint-level LCIA metrics. Along with any debate on the inclusion (or otherwise) of N_2O, there may be additional challenges of relevance. The calculation of ODP values is based on changes to *integrated global* ozone concentrations; however, the human health and environmental implications will be dependent on the *spatial distribution* of the ozone concentration changes. This relationship (between changing global concentration and the distribution of the changes) is fundamentally different for N_2O than for emissions of other ozone-depleting substances (Ravishankara et al. 2009). An obvious example of this is the negligible contribution made by N_2O to the extreme ozone layer thinning over polar regions. Further investigation is required on how best to include N_2O into the available endpoint LCIA models.

While there will clearly be some challenges in resolving the way forward on the issue of N_2O and ozone depletion, one of the strengths of LCA is its ability to accommodate such knowledge hurdles in a transparent way. As an example, the ReCiPe method provides up to three different sets of impact models for each environmental issue, with each set encapsulating a different approach to uncertainty at the science/policy interface. The first set typically has a shorter-term focus, the second occupies the middle ground and aims to reflect the most common policy principles, while the third takes a more precautionary perspective by including impact pathways that are considered less certain. Maybe it is not that big a leap to get scientific input on a *range* of ODP characterisation factors that could be included for N_2O emissions in LCA studies.

Doing so would then allow decision-makers to test the sensitivity of their LCA results to this issue, with the confidence that their approach reflects the best *available* scientific thinking. Not doing so risks the perverse outcome whereby the use of LCA would reinforce, rather than challenge, the common perspective that the ozone depletion problem has largely been 'fixed'. And since LCA will continue being used regardless—often to inform and justify decisions that have long-term ramifications—the interim inclusion of N_2O in LCA ozone depletion models would at least encourage decision-makers not to overlook this potentially significant issue.

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APPENDIX B1 CASE STUDY #1 - THE DIVERSE ENVIRONMENTAL BURDEN OF CITY-SCALE URBAN WATER SYSTEMS

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Lane, J.L., de Haas, D.W., Lant, P. (under review) The diverse environmental burden of city-scale urban water systems. *Water Research*

THE DIVERSE ENVIRONMENTAL BURDEN OF CITY-SCALE URBAN WATER SYSTEMS

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Abstract

Recent years have seen an increase in the use of Life Cycle Assessment (LCA) to inform urban water systems research. The attraction of LCA is its capacity to identify trade-offs across a broad range of environmental issues and a broad range of technologies. However, without some additional perspective on the scale of the results, prioritisation of these concerns will remain difficult. LCA studies at the whole-of-system level are required to identify the diversity of life cycle environmental burdens associated with urban water systems, and the main contributors to these impacts.

In this study, environmental impact profiles were generated for two city-scale urban water systems: one typical of many urban centres, with a high reliance on freshwater extraction and the majority of treated wastewater being discharged to the sea; and one that adopts a more diverse range of water supply and wastewater recycling technologies. The profiles were based on measured data for most system components, otherwise best available empirical data from the literature. Impact models were chosen considering the substantial methodological developments that have occurred in recent years.

System operations, directly within the sphere of influence of water system managers, play the dominant role in all but one of the 14 life cycle impact categories considered. While energy use is the main cause of changes in the impact profiles when the alternative water supply technologies are included, it is not the only important driver of impacts associated with city-scale urban water systems. Also extremely important are process emissions related to wastewater treatment and dams (notably fugitive gases, wastewater discharges, and biosolids disposal).

The results clearly indicate a diverse range of environmental impacts of relevance, extending beyond the traditional concerns of water use and nutrient discharge. Neither energy use, nor greenhouse gas footprints, are likely to be an adequate proxy for representing these additional concerns. However, methodological improvements will be required for certain LCA impact models to support future case study analysis, as will a comprehensive critique of the implications from selecting different impact models.

Keywords

life cycle assessment (LCA); urban water system; ozone depletion; greenhouse gas; phosphorus recovery; toxicity

1. Introduction

Urban water systems around the world are going through a period of substantial change (e.g. Brown et al. 2009, Marlow et al. 2013, Prosser 2011, Beck 2011). To reduce their dependence on extraction from natural freshwater systems, many cities are exploring alternative approaches to water supply. Wastewater systems are also increasing in complexity, driven by the push for water recycling, greater levels of nutrient removal, and stricter controls on the disposal of biosolids. At the same time, the urban water industry is being confronted with a

growing number of environmental management challenges – not the least of which include increasing energy demands, greater awareness of greenhouse gas (GHG) emissions, and concerns over contaminants in biosolids and wastewater reuse streams.

Life Cycle Assessment (LCA) methodology is sometimes used by the water industry to inform or critique such processes of change, because of its capacity to provide systematic and transparent assessment across a broad range of environmental issues. A defining feature of most such studies is that they are limited to only one element of the urban water system, comparing options targeted at a specific research or decision-making question. Many have compared choices to do with wastewater treatment or the handling of sewage sludge (see the literature surveys in Foley et al. 2010a, Rodriguez-Garcia et al. 2011, Remy and Jekel 2012, Corominas et al. 2013a, Yoshida et al. 2013). Fewer studies have critiqued different technologies for wastewater recycling (e.g. Muñoz et al. 2009, Pasqualino et al. 2011, Tangsubkul et al. 2005). Others have considered options for conventional drinking water treatment systems (see Friedrich 2002, Vince et al. 2008, Bonton et al. 2012, Igos et al. 2013), and alternative technologies such as seawater desalination or urban rainwater capture (e.g. Angrill et al. 2012, Mithraratne and Vale 2007, Hancock et al. 2012, Muñoz and Fernandez-Alba 2008, Godskesen et al. 2013, Del Borghi et al. 2013).

Invariably, options comparisons such as these reveal a number of trade-offs across the broad range of impacts considered under the LCA framework, and pose some challenging questions for decision makers. For example, a 10% reduction in effluent nutrient levels for a sewage treatment plant might come at the cost of a 5% increase in energy use. But does that cause a 5% increase in the organisation's overall GHG footprint, or a 0.5% increase? Might there be bigger opportunities for GHG mitigation in other components of the urban water system? Further, are GHG emissions the only externality worth considering, or are there other notable environmental pressures caused by this or other aspects of the whole system? Answering such questions would help decision-makers to prioritise across those environmental trade-offs, and management options, under consideration.

LCA applied across the entire urban water system can provide some of the perspectives required to overcome this decision-making complexity. While a small number of previous studies have already explored life cycle environmental burdens across full water and wastewater systems at the city scale (Amores et al. 2013, Lemos et al. 2013, Mahgoub et al. 2010, Lassaux et al. 2007, Lundie et al. 2004, Muñoz et al. 2010, Friedrich et al. 2009, Lundin and Morrison 2002, Slagstad and Brattebø 2014, Uche et al. 2013, Barjoveanu et al. 2014, see also the literature survey in Loubet et al. 2014), they will lack currency for many urban centres because of gaps in their scope. Most of those studies have excluded one or more of the contemporary technologies for water supply (e.g. rainwater tanks, seawater desalination, wastewater recycling) and wastewater treatment (e.g. advanced biological nutrient removal) technologies that are of particular interest in Australia and other parts of the world. Nor have those studies fully utilised the wealth of recent knowledge generated on operational aspects that were, historically, not well recognised. Examples include nitrous oxide emissions from wastewater treatment (Law et al. 2012), micropollutants in treated wastewater and biosolids, and energy use for small-scale infrastructure (Vieira et al. 2014).

Furthermore, renewed research effort into life cycle impact assessment models has led to a number of improvements in model fidelity being implemented or recommended in recent years (see overviews in Finnveden et al. 2009, Zamagni et al. 2012, Rack et al. 2013). Only a subset of these developments have been considered in the most recent LCA studies of integrated urban water systems (e.g. Amores et al. 2013, Lemos et al. 2013, Muñoz et al. 2010), with further analysis required to understand the challenges and opportunities arising from these advances in LCA methodology.

To address these gaps, this study combines contemporary scenarios and contemporary data, with a focus on key uncertainties in the LCA analysis of integrated urban water systems. The environmental burden of a city-scale, integrated urban water supply and wastewater system is

quantified, using the best available empirical data for modern water supply and wastewater technologies, and using life cycle impact assessment models across a broad range of issues related to environmental protection, resource availability and human health. The conclusions are developed in a number of ways: (i) two different system configurations are analysed, each with a very different mix of water supply technologies; (ii) the relevance of certain best-practice impact modelling approaches are considered in more detail; and (iii) the importance of certain inventory data assumptions is highlighted.

This work illustrates the diverse range of impacts that are relevant to the analysis of urban water systems, and the primary contributors in each case. It also provides quantitative data that could be used to benchmark other studies more focussed on specific sub-components of the urban water system.

2. Methodology

The functional unit for the study was defined as '*the provision of water supply and wastewater management services, for a one year period, to an urban population in the Gold Coast region of Australia'*. Reference flows were based on macro and micro-scale water and sewage balances created for each of the two scenarios under consideration, incorporating all water supply and wastewater management infrastructure in the urban area (Figure 1; Table 1; Table 2).

The modelling included the construction, assembly, operation and replacement of that infrastructure, as appropriate. The disposal of spent consumables (e.g. membranes) and infrastructure was excluded from the analysis, given the common finding that this life cycle stage makes a negligible contribution to the analysis of urban water systems (e.g. Bonton et al. 2012, Muñoz and Fernandez-Alba 2008, Uche et al. 2013, Raluy et al. 2006, Ortiz et al. 2007).

2.1 Scenario definition

The two scenarios encapsulate the technologies in use (or under consideration) in urban centres across much of the developed world (Figure 1; Table 1; and Table 2). Both scenarios are based on the infrastructure and urban footprint of the Gold Coast city in Australia. This particular region comprises many of the elements of interest to urban water system planners in other regions (e.g. advanced wastewater nutrient removal; large scale biosolids reuse; a diversified water supply system). Furthermore, it was possible to draw upon local empirical data for a number of operational aspects (e.g. residential water use profiles; direct GHG emissions) that are poorly characterised in most other urban centres.

Further details on the configuration, water balances, and technologies for each scenario are provided in the Supplementary Information.

The **'Traditional Infrastructure' scenario** represents the 'linear' model that is typical of urban centres in most developed countries. Mains supply is predominantly dam-sourced (83% of total supply), delivered from conventional drinking water treatment plants. Residential mains supply is supplemented with a small number of household rainwater tanks (2% of total supply). The wastewater system comprises centralised sewage collection and treatment, producing low nutrient effluent using a variety of bio-chemical nutrient removal processes. None of the treatment plants in question use anaerobic pretreatment, while anaerobic solids digestion and energy recovery are utilised for only 14% of the total sewage flow. Discharge of treated wastewater is to local waterways, with a portion reused for non-residential irrigation (15% of total water supply). All biosolids from the wastewater treatment are sent to agricultural reuse.

This infrastructure serviced 202,000 households in the year 2009, with the average population density taken as of 2.6 persons per household.

Water and sewage balances for the 'Traditional Infrastructure' scenario were determined by calibrating top-down and bottom-up estimates to derive assumptions for average water supply and demand. The top-down approach utilised city-scale water supply information (NWC 2009). The bottom-up approach was based on local empirical data for household water end-use (Willis et al. 2009, Willis et al. 2010) combined with rainwater tank yield modelling undertaken for this study.

The **'Diversified Infrastructure'** scenario incorporates a range of different urban water supply technologies, namely: increased supply from household rainwater tanks (15% of total supply); seawater desalination (29% of total); indirect potable wastewater recycling (10% of total); and non-potable direct wastewater reuse by the residential sector (1% of total). Non-residential, direct reuse of treated wastewater was scaled up to remain at 15% of the supply mix. While this scenario also incorporated a small increase in yield from the existing Gold Coast dams, the contribution from dam-based supplies is only 30% (down from 83%) of the total supply mix.

The modelled supply from each of these additional sources represents a high-end estimate of what might be possible in the Gold Coast context, given the technologies and policies in place at the time of this study. Combining all these together represents a substantial change from the status quo that existed in 2009. This 'extreme' scenario was used to explore whether the conclusions from this study would change with a very different water supply mix.

To enable a focus on the different water supply systems, all other changes to the system configuration of the 'Diversified Infrastructure' scenario were kept to a minimum. Accordingly, the following assumptions were used:

- The rate of per-capita water end-use was kept constant across both scenarios. Since the 'Diversified Infrastructure' scenario introduces a substantial increase in overall water supply capacity, the population base was increased (by ~2.7 times) so as to absorb that additional supply. The scale of this 'growth' is hypothetical, but is not unrealistic for the broader urban region that includes the Gold Coast City (see Supplementary Information section s1.2).
- The per-capita sewage characteristics (flow, COD, nutrient loads) were kept constant across both scenarios. The same overall wastewater management strategy (technology type; effluent and biosolids concentrations; effluent and biosolids discharge points) were used for both scenarios, other than for those changes induced by the adoption of two advanced wastewater recycling systems. This requires that the wastewater infrastructure stock and operations be scaled up linearly with scenario population, which effectively assumes that the expansion is achieved by replicating the existing stock (see section s2 of the Supplementary Information).

2.2 Inventory data collection

Operational data for each of the system component technologies were compiled from a range of sources, combining locally measured data with the best available sources of empirical and literature based information. Table 1 and Table 2 provide a summary of the key primary data sources for each of the component technologies. More in-depth information, and detailed inventory summaries, are provided in the Supplementary Information.

Treatment plant models for the 'Traditional Infrastructure' scenario were based on analysis of detailed monitoring data collected for the relevant treatment plants at the Gold Coast, for at least a 12 month period up to and including 2008. Where relevant, equivalent estimates for the 'Diversified Infrastructure' scenario were obtained by scaling up the model inventories in proportion to flow and/or population increases. System models for desalination and direct non-potable wastewater reuse were based on more recent monitoring data from Gold Coast systems that have been commissioned since 2009. Modelling of the indirect potable (wastewater) reuse system was based on detailed monitoring data from an equivalent system in a nearby region (O'Toole et al. 2008, Traves et al. 2008), adjusted for changes in influent

nutrient concentrations, and using distribution system designs (for product water and brine disposal) relevant to this study region.

Electricity use for household rainwater tank delivery pumps was estimated using empirical, end-use specific data from a range of recent Australian studies (Hauber-Davidson and Shortt 2011, Siems et al. 2013, Tjandraatmadja et al. 2013, Beal and Stewart 2011). A more detailed description of this calculation process, and background data, is provided in the Supplementary Information.

A detailed literature review was undertaken for direct emissions of N_2O, CH_4 and non-biogenic CO_2 from the water system infrastructure (see Supplementary Information section s3.4; table s9). Empirical evidence specific to the Gold Coast area was used to estimate CH_4 emissions from sewer systems (Foley et al. 2009, Guisasola et al. 2008, Liu et al. in prep) and from water supply reservoirs (Sherman et al. 2012, Grinham et al. 2011). Data from a neighbouring region, with similar urban characteristics, was used to quantify the influence of non-biogenic sewage carbon (Law et al. 2013). Other gas flux estimates were based on more generalised literature data that is less regionally specific.

Locally relevant industry data were used to estimate micropollutant concentration profiles for metals and organics in biosolids, metals in wastewater, and organics in recycled wastewater streams. For organic micropollutants in secondary treated wastewater, a hypothetical profile was developed using literature data from the local region (Reungoat et al. 2010, Watkinson et al. 2009).

Infrastructure construction data were in most cases taken from a variety of non-local sources (see the Supplementary Information for a full description), and annualised based on estimates of equipment lifespan. The exception was for the water and sewerage networks, where detailed piping inventories provided by the local Gold Coast water utilities were combined with assumptions on pipeline construction developed for a previous Australian study (Sharma et al. 2009).

Data for second-order inventories (e.g. power generation; materials supply) were taken, where possible, from the AusLCI (2012) and Life Cycle Strategies (Grant 2012) Australian LCI databases. Otherwise they were based on Ecoinvent data (Frischknecht et al. 2007). Electricity supply was modelled as the average of the generation mix in the state of Queensland, which for the inventories used in this study, is dominated by black coal (85% of total) and natural gas (12% of total) combustion plants.

2.3 Impact Assessment

This analysis follows the midpoint approach to Life Cycle Impact Assessment (LCIA). Midpoint indicators are measures of *potential* impact, defined at some intermediate point along the cause-effect chain from intervention (e.g. emission) to actual impact of concern (e.g. biodiversity; human health). While these midpoint-level indicators are necessarily incomplete in their description of environmental pressures, and do not aim to quantify actual damage, they often have the advantage of aligning well with metrics used for informing policy making.

The choice of 14 impact assessment models was tailored for relevance to urban water systems, in accordance with the principles specified in ISO14040 (2006). These were taken from a range of sources, giving consideration to the recommendations for European practice from a review of the best available models in 2009 (Hauschild et al. 2013), and our own review of more recent literature for the priority areas in this study. Models from the ReCiPe suite (Goedkoop et al. 2009) were adopted by default in four other cases, as this provides one of the most comprehensive, and extensively published, set of LCIA indicators available.

The final choice of impact assessment models is summarised in Table 3, with the Supplementary Information providing further details on model customisations applied specifically for this study.

Following the more general analysis across the 14 impact indicators (Sections 3.2 and 3.3), Section 3.4 provides a greater focus on the challenges associated with interpreting results for five indicators in particular – Stratospheric Ozone Depletion (OD), Ecotoxicity (MTX, FTX, TTX) and Minerals Depletion (MD). In each case, the critique considers whether the available models can adequately address 'emerging issues' under consideration by the urban water industry.

3. Results and Discussion

3.1 Operational inventories

Comparing across the different infrastructure components (Table 4) highlights that the 'new' water supply technologies, adopted as part of the 'Diversified Infrastructure' configuration, greatly increase the energy intensity of the overall system. For the 'Traditional Infrastructure' scenario, conventional sewage collection and treatment steps are the dominant power user (78% of total); due to the aeration demands for advanced nutrient removal, and high sewage and wastewater pumping requirements (see Table s11). Wastewater management makes a lesser contribution (33% of total) to the overall power consumption of the 'Diversified Infrastructure' scenario.

Conventional potable water treatment is the biggest driver of chemicals demand in these scenarios, with the contribution from sewage collection and treatment being much smaller (Table 4; also Table s10). This is despite the need for chemical nutrient removal applied to 23% and 30% of the total sewage flow of the 'Traditional' and 'Diversified' scenarios respectively. Our analysis did not, however, consider the intensive chemicals use for sewer odour and corrosion control that was implemented subsequent to the 2008 assessment period used for this study (see Ganigue et al. 2011).

Biosolids generated by the wastewater treatment process are the major solid waste stream under both scenarios (Table 4). The biosolids trucking distance (200km) defined for these scenarios is much lower than that required to transport many of the chemicals used, some of which are manufactured in cities 1000km (or greater) from the Gold Coast region. However, the volume of biosolids disposal is sufficiently large for this to represent the biggest life cycle demand for materials transport.

Sewage collection and treatment is also the major driver for direct gaseous emissions under both scenarios, particularly for N_2O (predominantly associated with treatment plant nutrient removal, and biosolids disposal), NH_3 (biosolids disposal) and direct CO_2 emissions (both from sewage treatment and biosolids disposal). CH_4 is the one exception since dam-sourced emissions are also substantial – for the 'Traditional Infrastructure' scenario, both this and sewer-sourced CH_4 emission levels are very similar.

3.2 Life cycle impact profile of the urban water systems

System operations, rather than infrastructure construction, dominate all but one of the life cycle impacts (Figure 2). This holds for both scenarios in this study, and is consistent with the findings of most other LCA studies into urban water system technologies (e.g. Corominas et al. 2013a, Bonton et al. 2012, Muñoz and Fernandez-Alba 2008, Lundie et al. 2004, Uche et al. 2013, Raluy et al. 2006). When the 'operational' impacts are broken down further, the significance of the conventional sewage management steps (collection and treatment) becomes clear. For the 'Traditional Infrastructure' configuration, these components represent the biggest operational contribution to most of the impact assessment category results (Figure 2).

3.2.1 Direct impacts

Pollutants derived from sewage inputs to the wastewater system make the majority contribution to the *Eutrophication* and *Ecotoxicity* results (Table 5). With all wastewater discharges in our two scenarios being to estuarine or coastal waterways, these are the major cause (respectively, 96% and 81% of total) of *Marine Eutrophication (MEU)* and *Marine Ecotoxicity (MTX)*. Residual contaminants in the land-applied biosolids are the dominant source (92% of total) of *Terrestrial Ecotoxicity (TTX)* potential, and also make a big contribution (40%) to the *Freshwater Ecotoxicity (FTX)* results. Nutrient loss to freshwater streams, via runoff or leaching from the biosolids applications, s the largest potential cause of *Freshwater Eutrophication (FEU)*, although the nutrient loss rates included in this analysis are conservative on the high side for Australian conditions (see Supplementary Information).

Freshwater Extraction Stress (FES) is one impact category where the role of wastewater systems is minor. Local water supply dam extractions dominate the life cycle *FES* results (Table 5); even with the relatively low WSI weighting factor assigned to the Gold Coast freshwater systems. According to the electricity supply inventories used in this study, the contribution from water 'embedded' in the generation of power supplied to water system operations is negligible (<1%).

Wastewater-related fugitive gas emissions make a notable contribution to a number of the impact results (Table 5). N_2O fluxes contribute 73-85% of the total *Ozone Depletion (OD)* results for the system, and 4-7% of the total *Global Warming (GW) score*. Dam-sourced CH_4 emissions also contribute 4% to the total *GW* of the 'Traditional Infrastructure' configuration, suggesting this issue might warrant more detailed consideration in future comparisons of water supply alternatives. Overall, direct gaseous emissions constitute 21% of total *GW* for the 'Traditional Infrastructure' system, and are far more substantial than the potential for biosolids carbon sequestration (−3%). Fugitive ammonia (NH_3) emissions make a large contribution to the results for *Terrestrial Acidification (TA)* and *Particulates Formation (PF)*, associated with the potential for NH_3 to initiate the secondary formation of NO_X and aerosols in the atmosphere.

These results suggest that analysis of fugitive gas emissions should be an important focus of future urban water studies. It will also be important to recognise the large uncertainties associated with estimating each of the emission sources. For example, if we had used an N_2O emission factor for wastewater treatment based on the average (15.5g-N_2O/kg-TN) rather than median (7.5g-N_2O/kg-TN) of available data, the direct emissions contribution increases to 26% of the total GW for the 'Traditional Infrastructure' scenario. Furthermore, the most important emission sources can all vary substantially across different locations. For example, dam-sourced CH_4 fluxes might be sensitive to climate, the extent of carbon input, nutrient status, and stratification behaviour (Sherman et al. 2012, Grinham et al. 2011, Barros et al. 2011, Gonzalez-Valencia et al. 2014). Large site-to-site variations are also expected for sewer-sourced CH_4 and sewage treatment N_2O emissions (see the Supplementary Information).

3.2.2 Indirect impacts

Of the *indirect* impacts occurring in the supply chain of operational inputs, the most notable are associated with electricity generation (Table 5). This is the single biggest life cycle contributor to *Fossil Fuel Depletion (FFD), Global Warming (GW), Photochemical Ozone Formation (POF), and Particulates Formation (PF)*. LCA results can vary substantially depending on the assumed electricity generation technology (Laurent et al. 2012, Turconi et al. 2013). Furthermore, using an historical electricity supply mix - such as done here, and in most LCA practice - provides only limited insight into how complex energy systems might respond to changes in demand, or how they might evolve over time. Given this, it may be important for future studies to consider a range of possible scenarios for defining the marginal electricity supply mix (Alvarez-Gaitan et al. 2014, Mathiesen et al. 2009).

Electricity use is not, however, the only important cause of peripheral life cycle impacts associated with this system. Along with the significant role of fugitive gaseous emissions noted

above, chemicals manufacturing and materials transport also make large contributions (up to 32% and 16% respectively) to the results for certain impact assessment categories (Table 5).

Furthermore, the results suggest that neither inventory-level accounting of electricity use, nor carbon footprinting, would make a reliable proxy (under all circumstances) for the indirect life cycle impacts associated with urban water systems. The relative contributions made by each supply chain (electricity; transport; chemicals; construction materials) vary substantially across the different impact categories (Table 5). In the case of water system scenario comparisons, it could not be assumed that a single metric (e.g. carbon footprint) would universally represent the differences across the broader spectrum of life cycle impacts associated with supply chain activities. This conclusion is consistent with the findings of an LCA impact review across ~4000 different products and services (Laurent et al. 2012), which highlighted the risks in using carbon footprints as the only metric for considering environmental sustainability concerns.

3.3 The implications of water supply system diversification

Implementation of non-traditional water supply technologies in Australia is, by and large, intended to reduce the intensity of freshwater use and nutrient discharge from the urban water system. For our 'Diversified Infrastructure' scenario, with its heavy reliance on such technologies, this is reflected in the *Freshwater Extraction Stress* and *Marine Eutrophication* results. On a per-household basis, these are 62% and 27% lower (respectively) than the equivalent results for the 'Traditional Infrastructure' configuration (Table 6).

However, adopting these 'new' technologies comes at the cost of substantial increases (+4% to +93%) in the intensity of many of the other impact category results (Table 6). Increased electricity usage is the main cause of these downsides, with the changes in chemicals manufacture and materials transport being of secondary importance to the difference between the two scenarios.

Despite this, the breakdown of the overall results is largely the same for the two scenarios (Table 5). There is little change to the toxicity profiles, which are primarily affected by process (wastewater and biosolids) contaminants. There is also little change in the nature of the life cycle human health impacts (*POF*, *PF*), which are dominated in both scenarios by power use and construction materials manufacture. Fugitive N_2O emissions are still the dominant source of *Ozone Depletion* potential, although the *Global Warming* from direct gas emissions becomes relatively small (10%) compared to that attributed to power generation (76% of total).

Even with the substantial increase in power intensity for urban water supply, sewage system operations (collection and secondary treatment) still make a major contribution to most of the life cycle impact category results of the Diversified Infrastructure scenario (Table 5). This conclusion might be expected to hold for other urban centres, given that the seawater desalination contribution (29% of the overall water supply mix) in our scenario is quite large. It is this desalination component that is responsible for most of the increased power consumption of the 'Diversified Infrastructure' system (Table 4).

3.4 The significance of impact model selection

In the available literature on LCA analysis of urban water technologies and systems, relatively few studies have considered whether the choice of impact models will affect the conclusions that might be drawn. In those that have, the discussion is mostly focussed on the implications of choosing between different 'off-the-shelf' packages[1] of impact models (Corominas et al. 2013a, Igos et al. 2013, Uche et al. 2013, Hospido et al. 2012, Zhou et al. 2011, Renou et al. 2008).

[1] In the LCA literature, these packages are more commonly termed 'impact assessment methods'.

There is some risk in relying on these past comparisons to inform the design of future analytical studies. Along with 'upgrades' of the most commonly used model packages[2], there have been substantial developments in more specific areas of life cycle impact assessment (LCIA) methodology over recent years (see Finnveden et al. 2009, Zamagni et al. 2012, Rack et al. 2013, Hauschild et al. 2013). However, not all these developments have been incorporated into the commonly used 'off-the-shelf' models. Furthermore, while the importance of incorporating local hydrological and/or ecological sensitivity to water withdrawals or nutrient discharges has been recognised (Corominas et al. 2013b, Godskesen et al. 2013, Amores et al. 2013, Muñoz et al. 2010, Gallego et al. 2010), the significance of many other developments has not yet been considered in the context of quantitative urban water systems analysis.

Here, we focus on three other 'environmental' issues that are already on the radar of the urban water industry – N_2O emissions, organic micropollutants in wastewater and biosolids, and phosphorus resource recovery. In each case, our results illustrate that the conventional impact model choice does not conform to contemporary scientific focus in that particular discipline.

3.4.1 N_2O emissions and the ozone layer

In addition to the well-recognised global warming implications, N_2O emissions are expected to be the major anthropogenic source of stratospheric ozone depletion into the foreseeable future (Kanter et al. 2013, Portmann et al. 2012). Regardless of whether or not this translates into a policy response by the international community, one of the key roles for LCA is to broaden the horizon of decision makers beyond those imposed by institutional settings. We have therefore followed the rationale that if LCA studies are to include metrics for ozone layer depletion, then N_2O should be a part of those considerations (Lane and Lant 2012).

The implications of this methodological choice can be illustrated by comparing the results for the 'Traditional' scenario (as per Table 5) against those that would be obtained using a more conventional approach to *OD* assessment (Table 7). Excluding N_2O from the impact model gives the impression that water system managers have relatively little control over their contribution to this environmental issue, as the other life cycle emissions of ozone depleting substances are dispersed across various supply chain activities. But factoring in the role of N_2O leads to the opposite conclusion, since direct N_2O emissions from the wastewater system dominate the life cycle *OD* results.

3.4.2 Toxicity of organic micropollutants

Direct process flows make substantial contributions to the life cycle *Marine*, *Freshwater* and *Terrestrial Ecotoxicity* results for both scenarios (Table 5), primarily associated with residual metals in the treated wastewater and sewage sludge streams (Table 8). In all cases, the equivalent contributions from residual organic micropollutants (e.g. pharmaceuticals and pesticides) are negligible. This contrasts with the research agenda of the water industry in recent years, which has put a high priority on concerns over organics in wastewater and biosolids.

This disparity suggests that improvements in the fidelity of LCIA toxicity models may be required, if they are to provide robust analysis of urban water process flows. Their shortcomings for assessing metals have been widely discussed (Corominas et al. 2013a, Ligthart et al. 2004, Diamond et al. 2010), with recent work highlighting that conventional approaches can overstate the implications of metals emissions, by an order of magnitude or

[2] For example, Impact2002+ (Jolliet et al. 2003) transforming into IMPACT World+ (2014); ReCiPe (Goedkoop et al. 2009) building on the foundations of CML-IA (Guinée et al. 2002) and EcoIndicator99 (Goedkoop and Spriensma 2000); and CML-IA itself being regularly updated, most recently in 2013 (CML 2013).

more, under some conditions (Gandhi et al. 2011, Gandhi et al. 2010). Furthermore, studies using novel bioanalytical approaches have shown that additive, chemical-specific toxicity estimates could understate the implications of residual wastewater organics by an order of magnitude or more (e.g. Escher et al. 2011). Those studies highlight the difficulty in identifying and measuring all possibly relevant organic species, and the constraint this imposes on the utility of the chemical-specific approach to toxicity quantification. This same concern is relevant to conventional LCA practice, which also employs a chemical-specific analytical framework. In conclusion, ascertaining the relative importance of metal vs. organic pollutants is not possible with the LCIA toxicity models used here.

These concerns do not necessarily, however, invalidate the conclusion that wastewater system flows make a substantial contribution to the overall toxicity burden of the water system life cycle. With the models used here, supply chain toxicity contributions are also dominated by metals emissions (Table 8), and might therefore also be affected by any change in model bias between metals and organics. A more comprehensive evaluation would be required to determine whether such a recalibration would make sewage pollutants appear more, or less, significant from a life cycle perspective.

3.4.3 The significance of phosphorus recovery

In light of global concerns about escalating demand for finite mineral phosphorus resources, many urban water industry stakeholders are looking to capitalise on opportunities to recover nutrients from the wastewater system (Sartorius et al. 2012). There is a long history in LCA research of using metrics to combine and compare the depletion of different mineral types, providing insight into the priority concerns and biggest life cycle contributions. This suggests that LCA should have a useful role to play in assessing the significance of wastewater phosphorus recovery.

However, we suggest there is too much inherent uncertainty in the available LCA modelling approaches, for them to provide robust guidance on this topical issue. Using our default choice of impact model for *Minerals Depletion (MD)*, the conclusions from this case study contrast starkly with the water industry's enthusiasm for phosphorus recovery. The 'Traditional Infrastructure' scenario involves 77% of sewage phosphorus being returned to agricultural fields, yet the associated fertiliser offsets reduce the overall *Minerals Depletion* score by less than 1%. According to the metric used here, the *Minerals Depletion* associated with the supply of construction materials, chemicals and electricity is far greater; implying that reducing or substituting other system inputs could make a more important contribution to the goal of long term resource sustainability. Changing the assumed displacement ratio for diammonium phosphate (DAP) fertiliser would have no bearing on this conclusion.

This discrepancy highlights the importance of paying careful attention to the impact models that are chosen for LCA studies applied to urban water systems. Our choice of *Minerals Depletion* impact model was based on European Commission recommendations (Hauschild et al. 2013, ECJRC 2011), and is widely used in mainstream LCA practice. However, its utility for use in LCA resource depletion assessment has been challenged, on the grounds that it takes too simplistic a view of resource scarcity (Vieira et al. 2012, Klinglmair et al. 2014, Schneider et al. 2014). More recently developed methodologies attempt to introduce a stronger focus on the economic (rather than physical) implications of resource scarcity (Goedkoop et al. 2009, Vieira et al. 2012, Schneider et al. 2014), and might therefore go some way to accommodating the sense of urgency in the phosphorus supply-demand debate. Unfortunately they do not include impact factors for phosphate resources, hence are not able to shed light on the significance, or otherwise, of sewage phosphorus recovery.

4. Conclusions

Analysis of a large scale, integrated urban water supply and wastewater system shows that wastewater management plays a significant role in its overall environmental burden, even when that system is based on very energy intensive water supply technologies. This wastewater system contribution is stronger than in many similar studies, at least in part due to the more comprehensive coverage of important wastewater system inventories (e.g. fugitive GHG emissions) utilised here.

Electricity use is an important driver of indirect environmental impacts across the life cycle. While it represents a somewhat extreme change, the more diverse water supply mix considered in this study does substantially increase (by 2.3-fold) the overall energy intensity of the system. Those alternate water supply approaches reduce the potential environmental pressures associated with freshwater extraction (by 62%) and nutrient discharge (by 27%), but this comes at the expense of large increases (up to 93%) in many other life cycle environmental impacts.

While electricity use strongly influenced this scenario comparison, our results suggest that measuring electricity consumption will not make a reliable proxy for managing the greenhouse gas (GHG) burden of the overall urban water system. Even with a fossil fuel-intensive power supply mix, direct gas emissions (predominantly N_2O and CH_4) from dams and the wastewater system contribute up to 21% of the total GHG footprint for our scenarios. It is conceivable that this contribution could be far greater in some cases, given our estimates for the most substantial direct GHG fluxes could be conservative for some other urban centres. Water utilities will require improved data on these process gas emissions, if they are to understand the true GHG risks and opportunities inherent in their infrastructure systems.

It should also be recognised that GHG accounting will not be sufficient to recognise, nor help manage, other environmental externalities that may challenge the industry. Urban water process flows make substantial contributions to other impact categories that are only weakly affected by the electricity supply chain. Stratospheric ozone depletion (caused by N_2O emissions) and ecotoxicity impacts (associated with sewage micropollutants) are two such possibilities.

Given the number and complexity of issues that are important to the LCA results, future urban water systems analysis will require substantial rigour applied when estimating the scale of these key, but less well understood, process flows. Analysis based primarily on water, electricity, chemicals and nutrient inventories would not do justice to the wealth of other process information (e.g. fugitive GHG emission rates; water, wastewater and biosolids micropollutant concentrations) that has been collected over the last decade. While the compilation of detailed empirical and literature based data might be used to inform other such studies, further research is needed to understand and reduce some of the considerable uncertainties that remain. The potential for large variations across different locations and/or case studies should not be overlooked.

Notwithstanding these uncertainties, it is clear that many of the key drivers of life cycle impacts are directly within the sphere of influence of urban water system managers. It is also apparent that pressures on local aquatic systems are important, even when considered from the broader life cycle perspective. Focussed risk assessment should therefore remain an important step in urban water decision making, when there are local environments directly exposed to impacts from water system activities.

LCA does have an important role to play in urban water systems analysis, as it provides a robust framework for quantitatively addressing (a) the more global of these ‘direct’ impacts, along with (b) those more indirect drivers where the impacts are likely to be felt at other points in time or space. This could prove increasingly useful, as the urban water industry extends its focus beyond the traditional concerns of microbial human health protection, financial cost, water use and nutrient discharge.

However, a greater focus on the fidelity of existing LCA impact models is required, if practitioners are to recognise key limitations in their analysis of urban water systems. For example, if future LCA studies on urban water systems are to include the issue of stratospheric ozone depletion, then the importance of fugitive process emissions should not be overlooked. In the contrasting case of toxicity and minerals resource assessment, further improvements to the available LCA impact models may be required before credible analysis is possible. For LCA based analysis to add value to the urban water system planning process, careful attention should be paid to the choice of impact models that are used.

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Figure Captions

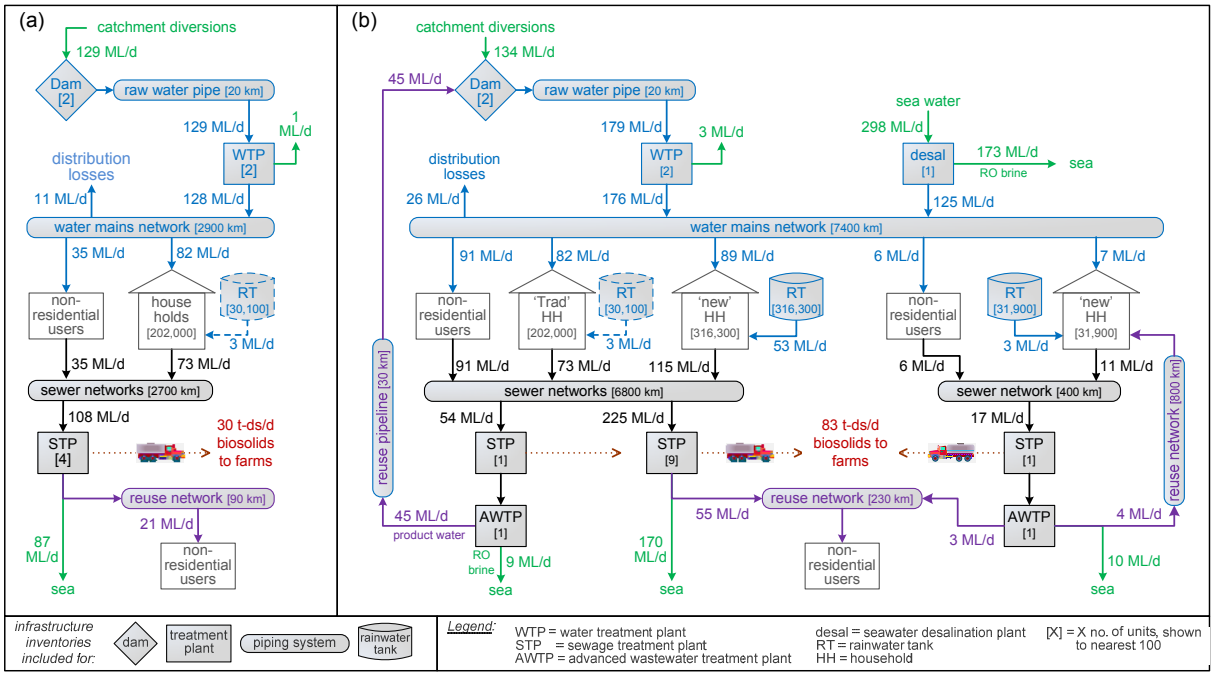

Figure 1: Flowsheets for the (a) 'Traditional Infrastructure' scenario, and (b) 'Diversified Infrastructure' scenarios, showing the main process flows and the number of units [X] for each system component. The difference between total water use, and total water entering the sewer system, is the water used for irrigation and other external purposes – see Table 4.

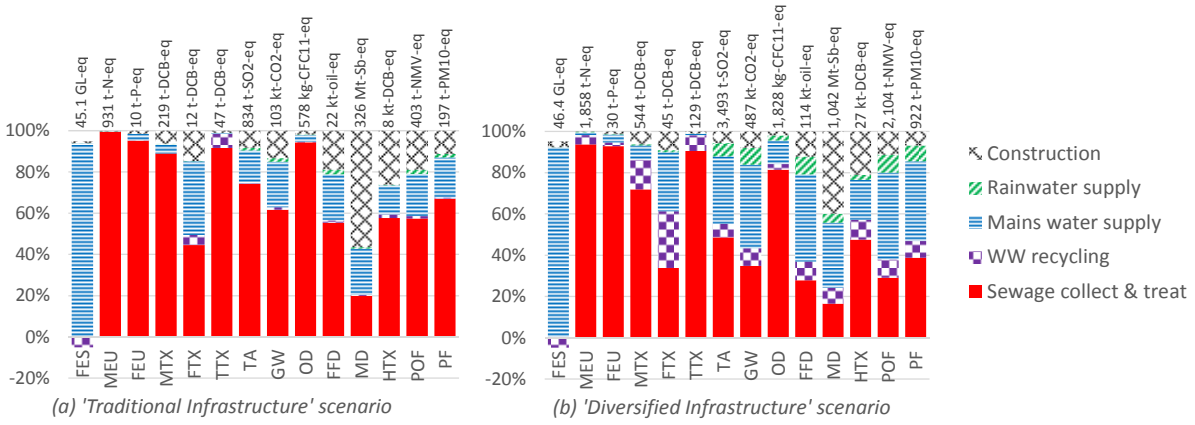

Figure 2: Impact results for the (a) 'Traditional Infrastructure' & (b) 'Diversified Infrastructure' scenarios. For each impact category, the absolute (annualised) results are provided, along with a breakdown (as % contributions) of the overall Impact Indicator results by water system component.

Table Captions

Table 1: Information sources for modelling the components of the 'Traditional Infrastructure (T)' scenario. More detailed descriptions are provided in the Supplementary Information.

Table 2: Information sources for modelling the components of the 'Diversified Infrastructure (D)' scenario. More detailed descriptions are provided in the Supplementary Information. (T) refers to the stock of the 'Traditional Infrastructure' scenario (as per Table 1).

Table 3: Impact categories, and impact models, used for this study. Customised impact models are described in more detail in the Supplementary Information.

Table 4: Summary of inventory estimates for each of the infrastructure components, with an overall GHG intensity for operations of each system component. Results are expressed per-household (hh), to allow direct comparison across the different scaled scenarios. Both scenarios have the same average population density of 2.6 persons/hh.

Table 5: Impact results for the Traditional & Diversified scenarios, separated into contributions from direct process flows vs. those associated with the supply chain of operational inputs. Contributions are only shown if they are ≥1% in a positive or negative direction. WW = wastewater.

Table 5: Comparison of the scenarios on a per-household (hh) basis, showing (a) overall change in the per-hh results from the 'Traditional Infrastructure' (T) and 'Diversified Infrastructure' (D) scenarios; and (b) the system components that make the biggest contribution to those overall changes. For each of the items shown, the contribution was calculated as (D-T)/ ΣT. This represents the change in impact result (D-T) for each of the items, divided by the total impact calculated for the 'Traditional Infrastructure' scenario (ΣT). Contributions are only shown if >=1% in a positive or negative direction.

Table 6: The effect of impact model choice on OD results for the 'Traditional Infrastructure' scenario. The first set of results employs a conventional LCA impact model for ozone depletion, including only halocarbon emissions in the analysis. The second set of results incorporates the additional impact factor for anthropogenic N_2O emissions (as per Table 5: Impact results for the Traditional & Diversified scenarios, separated into contributions from direct process flows vs. those associated with the supply chain of operational inputs. Contributions are only shown if they are ≥1% in a positive or negative direction. WW = wastewater.).

Table 7: 'Traditional Infrastructure' scenario results for (a) Marine, (b) Freshwater, and (c) Terrestrial Ecotoxicity, showing the relative contributions (in %) from metals and organics, for process streams and for 'indirect' impacts occurring the in supply chain of operational inputs to the urban water system.

Table 1: Information sources for modelling the components of the 'Traditional infrastructure (T)' scenario. More detailed descriptions are provided in the Supplementary Information.

Comp-onent	Infra-structure[a]	Reference flows	Measured operations data[b]	Inferred operations data
Dams & potable treatment	Based on 2008 infra-structure	Matched to 2008 data on WTP production, residential use, & non-residential use	Water flows; Sludge flows & disposal; Chemicals, Fuel & Total Electricity usage.	CH_4 emissions based on literature data for the 2 major Gold Coast water-supply dams (Sherman et al. 2012, Grinham et al. 2011). Calculated sludge metal (Al, Fe) concentrations.
Potable water distribution			Distribution losses; Electricity usage.	
Rainwater tanks	Estimated tank numbers in 2008	Household scale rainwater yield modelled with TANK software (Vieritz et al. 2007).		Yield modelling based on the default end use profile (Table s3); and locally specific assumptions for supply side configuration. Mains water backup configurations based on anecdotal local information. Electricity usage for rainwater supply based on data from Australian field and laboratory studies (Hauber-Davidson and Shortt 2011, Siems et al. 2013, Tjandraatmadja et al. 2013).
Sewage collection	Based on 2008 infra-structure	Total throughput & concentration profiles (N,P,COD) matched to 2008 STP operations data. Residential vs. non-residential flow split inferred through system balance.	Electricity usage.	Locally relevant literature data for non-biogenic sewage carbon (Law et al. 2013) & dissolved CH4 generation (Foley et al. 2009, Guisasola et al. 2008, Liu et al. in prep).
Sewage treatment			Flows & concentrations (N, P, COD, metals) for sewage and treated wastewater; Biosolids quantity, water content, micropollutant concentrations (metals, organics) & transport distance; Chemicals, Fuel & total Electricity usage.	Organic micropollutant concentrations in secondary treated wastewater based on local literature data (Reungoat et al. 2010, Watkinson et al. 2009). Biosolids nutrient and carbon content based on mass balances, with fertiliser displacement calculated using bioavailability factors from Foley et al. (2010a). Fugitive gas emissions based on mass balance and literature informed estimates for N_2O, NH_3, CH_4 & CO_2 (Foley et al. 2010a, Aboobakar et al. 2013, Ahn et al. 2010, Daelman et al. 2013, Foley et al. 2010b, Joss et al. 2009, Kampschreur et al. 2008, Ni et al. 2013, Sun et al. 2013, Ye et al. 2014, GWRC 2011, Brown et al. 2010, de Haas et al. 2009, Foley and Lant 2007).

[a] *All based on infrastructure in place at the Gold Coast city*

[b] *Data (for a period of at least 12 months) collected for the Gold Coast infrastructure included in this study*

Table 2: Information sources for modelling the components of the 'Diversified Infrastructure (D)' scenario. More detailed descriptions are provided in the Supplementary Information. (T) refers to the stock of the 'Traditional Infrastructure' scenario (as per Table 1).

Comp-onent	Physical infra-structure[a]	Reference flows	Measured operations data[b]	Inferred operations data
Dams	2011 infrastructure	Environmental withdrawals set at long term planning yield, supplemented with IPR product water.		CH_4 emissions as per Table 1
WTP	no change to (T)[c]	Throughput scaled up to match increased dam withdrawals.	2008 data as per Table 1[d]	Inputs & outputs scaled up with increased throughput.[d]
Potable water distribution	Piping of (T) scaled up linearly with population increase			
Seawater desal-ination	2011 infrastructure	Assumed to run at full capacity.	Electricity usage for treatment and discharge of product water (Poussade et al. 2011).	Chemicals usage based on review of literature and unpublished data (Muñoz and Fernandez-Alba 2008, Leslie 2010, Mrayed and Leslie 2009). Flow & electricity usage (Hall et al. 2011) for downstream booster pumps in the distribution system.
Rainwater tanks	Tank stock of (T); plus 1 tank for each new household added with this scenario.	As per Table 1		
Sewage collection	Piping of (T) scaled up linearly with population increase	Throughput scaled up to the capacity of the infrastructure in place in 2009.	2008 data as per Table 1	Inputs & outputs as per Table 1, scaled up in proportion to population increase.
STP & WW recycling (Direct Non-potable Reuse - DNR)[e]	2009 infrastructure for the STP & AWTP; and for the recycled water distribution network, scaled up linearly with the modelled population increase.		Flows & concentrations (N,P,COD) for sewage, STP effluent & AWTP product water; Biosolids quantity, water content, micropollutant concentrations (metals, organics) & transport distance; Chemicals, fuel & total electricity usage for STP & AWTP.	Energy, chemicals & waste flows estimated for the scenario conditions, using a simplified set of linear models derived from the available STP & AWTP data;[d] Organic micropollutant concentrations in STP effluent - as per Table 1; Assumed negligible organics removal in the AWTP. Other STP inputs/ outputs estimated as per Table 1.
All other Sewage treatment	2008 infrastructure (as per the 'Traditional' scenario), scaled up linearly with population increase	Sewage & treated wastewater flows increased linearly with population increase. Concentrations (COD, N, P) profile as per Table 1.	2008 data as per Table 1	Inputs & outputs as per Table 1, scaled up in proportion to population increase. Organic micropollutant concentrations as per Table 1.
WW recycling (Indirect Potable Reuse - IPR)[e]	2011 infrastructure in a neighbouring region.	Feed flow & concentrations (N, P, TOC) matched to effluent from largest STP in place in 2008; Product water recovery rate & concentrations (N,P,TOC) taken from operating plant data.	Flows & concentrations (N, P, TOC) for feed, product water & brine; Sludge quantity & transport distance; Electricity (Poussade et al. 2011), Chemicals & Fuel usage. Removal ratios for organic micropollutants.	Energy, chemicals and waste flows estimated for the scenario conditions, using a simplified set of linear models derived from the available AWTP data.[d] N_2O emissions based on calculated mass balance and an emission factor as per STP modelling in Table 1.

[a] *Based on infrastructure in place at the Gold Coast city, unless otherwise noted*

[b] *Data provided by infrastructure operators, unless otherwise noted*

[c] *The two WTP's in place in 2008 had sufficient spare capacity to absorb the increased throughput set for this scenario.*

[d] *Estimates were made of the flow-dependent & time-dependent contributions to the operations of each infrastructure item, and taken into account when scaling the available empirical operations data.*

[e] *Data provided by the water utility was for throughput & influent conditions that differed from those used in the scenario design*

Table 3: Impact indicators, and impact models, used for this study. Customised impact models are described in more detail in the Supplementary Information.

Indicator	Proxy for…	Model used
Freshwater Extraction Stress (FES)	Ecosystem impacts from disruptions to the hydrological cycle	Extractions weighted by source catchment sensitivity using Water Stress Index (WSI) values of Pfister et al. (2009), with a generic Australian value used for supply chain water use – *see Supplementary Information.* Rainwater tank interceptions in the Gold Coast region were assumed to have no freshwater hydrological effect (WSI = 0).
Freshwater Eutrophication (FEU)	Ecosystem impacts from nutrient enrichment and oxygen depletion in waterways	Wastewater discharge & irrigation losses in Gold Coast region, reach estuarine or coastal receiving waters sensitive to both N and P inputs. All other nutrient emissions reach P-limited freshwater streams or N-limited marine waters following the ReCiPe approach (Goedkoop et al. 2009, Struijs et al. 2011). Primary oxidation was accounted for in all cases (Karrman and Jonsson 2001) – *see Supplementary Information.*
Marine Eutrophication (MEU)		
Marine (MTX), Freshwater (FTX), & Terrestrial (TTX) Ecotoxicity	Ecosystem and human health impacts of chemical emissions to air, water and land	Midpoint versions of the USES-LCA 2.0 toxicity modelling package (van Zelm et al. 2009) were used to assess metal and organic contaminants in wastewater effluents, biosolids, and supply chain emissions (to air, water & land).
Human Toxicity (HTX)		
Terrestrial Acidification (TA)	Ecosystem impacts from soil acidification	Models taken from the ReCiPe suite (Goedkoop et al. 2009, van Zelm et al. 2008, Van Zelm et al. 2007)
Photochemical Ozone Formation (POF)	Human health impacts from urban smog (POF), & inhalation of particulate matter (PF)	
Particulates Formation (PF)		
Global Warming (GW)	Human & ecosystem impacts from climate change (GW), & depletion of the stratospheric ozone layer (OD)	100-year direct Global Warming Potential (GWP) factors from IPCC (2007); with adjustments for soil sequestration of carbon, fossil & biogenic CH_4 including secondary effects from oxidation to CO_2 (Munoz et al. 2013), & secondary N_2O as a result of NH_3 volatilisation (IPCC 2006)
Stratospheric Ozone Depletion (OD)		Ozone Depletion Potential (ODP) factors from Daniel et al. (2011) for halocarbons, and Lane et al (2012) for N_2O – *see Supplementary Information.*
Fossil Fuels Depletion (FFD)	Resource availability for future generations	Model taken from the ReCiPe suite (Goedkoop et al. 2009).
Minerals Depletion (MD)	Resource availability for future generations	'Economic Reserve' version of the CML-IA v4.2 model for Abiotic Resource Depletion (CML 2013, van Oers et al. 2002) – *see Supplementary Information.*

Table 4: Summary of inventory estimates for each of the infrastructure components, with an overall GHG intensity for operations of each system component. Results are expressed per-household (hh), to allow direct comparison across the different scaled scenarios. Both scenarios have the same average population density of 2.6 persons/hh.

		Traditional infrastructure configuration (202,000 hh)				Diversified infrastructure configuration (550,235 hh)						
		Dam & WTP	Rain tanks	overall Water Supply	overall Sewage Mgmt	Dam & WTP	Rain tanks	Desal	IPR	DNR	overall Water Supply	overall Sewage Mgmt
Flows - supply												
Product water [a]	(ML/d)	128	3[c]			176[e]	59	125	45	17		
Yield to users [b]	(ML/d)	117	3[d]	120	21	161[e]	59	114	41	7	342	55
Flows - by use												
internal uses - to sewer	(ML/d)			107							293	
external [f] - not to sewer	(ML/d)			13	21						49	55
Flows - to sewer												
from water users	(ML/d)				107							293
WTP waste	(ML/d)				1							3
Electricity use												
Total	(Wh/hh/d)	167	26	193	700	99	203	872	153	22	1,350	672
Chemicals supply												
Usage	(g/hh/d)	117	--	117	28	59	--	39	39	12	149	34
Transport	(kg.km/hh/d)	46	--	46	21	23	--	27	27	1	79	24
Direct gas emissions												
NH_3	(g-NH_3/hh/d)	--	--	--	1.9	--	--	--	--	<0.01	<0.01	1.9
N_2O	(g-N_2O/hh/d)	--	--	--	0.4	--	--	--	<0.01	<0.01	<0.01	0.4
CH_4	(g-CH_4/hh/d) [g]	2.5 / 0	--	2.5 / 0	2.9 / 0.3	1.1 / 0	--	--	--	--	1.1 / 0	2.8 / 0.3
CO_2	(g-CO_2/hh/d) [g]	--	--	--	344 / 38	--	--	--	1.0	0.2	1.3	339 / 38
C sequestered	(g-C/hh/d) [g]	--	--	--	11 / 1.2	--	--	--	--	--	--	11 / 1.3
Byproducts generated												
Biogas	(L/hh/d)	--	--	--	15	--	--	--	--	--	--	14
Sludge	(wet g/hh/d)	48	--	48	15	22	--	--	47	0.7	70	15
Sludge transport	(wet kg-km/hh/d)	--	--	--	0	--	--	--	0.9	0.1	1	0
Biosolids	(dry g/hh/d)	--	--	--	147	--	--	--	--	0	0	150
Biosolids transport	(wet kg.km/hh/d)	--	--	--	261	--	--	--	--	0	0	262
Products displaced												
Stream water use [h]	(L/hh/d)	--	--	--	-35	--	--	--	--	-2	-2	-11
Fertiliser use	(g/hh/d)	--	--	--	-21	--	--	--	--	0	0	-21
Fertiliser transport	(kg.km/hh/d)	--	--	--	-1	--	--	--	--	0	0	-1
GHG footprint	(g-CO2e/hh/d)	308	25	333	906	166	197	818	171	23	1,375	888
	(t-CO2e/ML) [i]	0.53	1.79	0.56	1.69	0.57	1.82	3.93	2.28	1.75	2.21	1.65

[a] *Incorporates all product water*

[b] *Net of any distribution losses*

[c] *Includes backup mains water delivered via tank topup*

[d] *Rainwater yield only (excludes all backup water)*

[e] *Includes IPR product water*

[f] *e.g. garden watering, parks irrigation, etc…*

[g] *Values shown as 'X / Y' represent the flows containing biogenic / non-biogenic carbon*

[h] *Local stream extractions delivered directly to users, rather than via the mains water system*

[i] *GHG footprint (t-CO_2-e) per ML-yield (for water supply); & per ML-sewage (for sewage management)*

Table 5: Impact results for the Traditional & Diversified scenarios, separated into contributions from direct process flows vs. those associated with the supply chain of operational inputs. Contributions are only shown if they are ≥1% in a positive or negative direction. WW = wastewater.

		FES (%)	MEU (%)	FEU (%)	MTX (%)	FTX (%)	TTX (%)	TA (%)	GW (%)	OD (%)	FFD (%)	MD (%)	HTX (%)	POF (%)	PF (%)
Direct	dam extractions	104/101													
	WW to waterways[a]		96 / 94	0 / 1	81/76	0 / 19							3 / 3		
	WW to irrigation[a]	-6 / -6				5 / 4	7 / 7				-1 / 0		3 / 4	-1 / 0	
	biosolids transport + use[a]		3 / 5	201/196	2 / 2	40 / 31	92 / 92					0 / 0	61 / 51		
	avoided fertiliser use[a]		-2 / -3	-106/-104		-1 / 0	-1 / -1					0 / 0	-28 / -24		
	Dams - CH_4								4 / 1						
	WW system - CH_4[b]								6 / 3					1 / 0	
	WW system - CO_2[b]								3 / 2						
	WW system - N_2O[b]								7 / 4	80 / 70					
	WW system - NH_3[b]		1 / 2					41 / 27	1 / 0	6 / 5					23 / 14
	WW system – other							1 / 1						1 / 1	1 / 1
	WW system -C sequest'n[b]								-3 / -2						
Inputs	electricity use		1 / 3	1 / 2	2 / 6	6 / 11	0 / 1	41 / 61	59 / 77	12 / 23	69 / 82	22 / 42	11 / 22	70 / 83	55 / 73
	waste transport				6 / 6	2 / 2		1 / 1	2 / 1		4 / 2		16 / 14	3 / 1	2 / 1
	chemicals supply	0 / 1		3 / 3	2 / 3	32 / 24		8 / 5	7 / 4		8 / 4	22 / 18	7 / 7	7 / 4	8 / 5
Construction materials		1 / 3		2 / 1	6 / 6	15 / 9	1 / 1	8 / 6	13 / 8	2 / 2	19 / 12	56 / 40	26 / 21	19 / 11	11 / 7
other						0 / 1			1 / 1				0 / 1		
Total		100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%

[a] *excluding direct greenhouse & NH_3 gas emissions from the disposal site*

[b] *including direct greenhouse & NH_3 gas emissions from the disposal of wastewater and biosolids*

Table 6: Comparison of the scenarios on a per-household (hh) basis, showing (a) overall change in the per-hh results from the 'Traditional infrastructure' (T) and 'Diversified infrastructure' (D) scenarios; and (b) the system components that make the biggest contribution to those overall changes. For each of the items shown, the contribution was calculated as (D-T)/ ΣT. This represents the change in impact result (D-T) for each of the items, divided by the total impact calculated for the 'Traditional infrastructure' scenario (ΣT). Contributions are only shown if >=1% in a positive or negative direction.

(a)

	FES	MEP	FEP	MTX	FTX	TTX	TA	GW	OD	FFD	MD	HTX	POF	PF
	kL-eq	kg-eq N	g-eq P	g-eq DCB	g-eq DCB	g-eq DCB	kg-eq SO2	kg-eq CO2	g-eq CFC11	kg-eq oil	t-eq Sb	kg-eq DCB	kg-eq NMV	kg-eq PM10
'Traditional infrastructure'	223	4.6	52	1,082	62	231	4	512	2.9	108	1,614	41	2.0	1.0
'Diversified infrastructure'	84	3.4	54	990	81	234	6	885	3.3	208	1,894	49	3.8	1.7
Change	-139 (-62%)	-1.2 (-27%)	+2 (+4%)	-93 (-9%)	+19 (+31%)	+3 (+1%)	+2 (+54%)	+373 (+73%)	+0.4 (+16%)	+100 (+93%)	+280 (+17%)	+8 (+19%)	+1.8 (+92%)	+0.7 (+71%)

(b)

		FES	MEP	FEP	MTX	FTX	TTX	TA	GW	OD	FFD	MD	HTX	POF	PF
Direct	dam extractions	-66%													
	WW --> water [a]		-28%	+1%	-12%	+25%							+1%		
	WW --> irrigation [a]	+4%											+2%		
	biosolids transport + use [a]			+4%											
	avoided fertiliser use [a]			-2%											
	direct gases – dams								-2%						
	direct gases - WW system [b]														
Inputs	electricity use		+1%	+1%	+3%	+8%		+52%	+75%	+15%	+88%	+28%	+15%	+89%	+70%
	waste transport														
	chemicals supply												+2%		
Construction materials							-3%					+5%	-10%	-1%	+3%
other							+1%	+1%	+2%		+1%				
Total			-62%	-27%	+4%	-9%	+31%	+1%	+54%	+73%	+16%	+93%	+17%	+19%	+92%

[a] *excluding direct greenhouse & NH_3 gas emissions from the disposal site*

[b] *including direct greenhouse & NH_3 gas emissions from the disposal of wastewater and biosolids*

Table 7: The effect of impact model choice on OD results for the 'Traditional infrastructure' scenario. The first set of results employs a conventional LCA impact model for ozone depletion, including only halocarbon emissions in the analysis. The second set of results incorporates the additional impact factor for anthropogenic N_2O emissions (as per the results in Table 5).

		excluding N2O	including N2O
Total OD (kg-CFC11-eq/y)		1.0	578
contributions	WW system N2O	--	86%
	electricity generation	1%	12%
	chemicals manufacture	53%	1.0%
	transport (biosolids, chemicals)	0%	0.2%
	infrastructure construction	46%	1.8%

Table 8: 'Traditional infrastructure' scenario results for (a) Marine, (b) Freshwater, and (c) Terrestrial Ecotoxicity Potential, showing the relative contributions (in %) from metals and organics, for process streams and for 'indirect' impacts occurring the in supply chain of operational inputs to the urban water system.

(a)	wastewater disposal	biosolids disposal	supply chain activities	Total
Metals	75	1.3	22	98
Organics	1.3	0.03	0.00	1.3
Total	77	1.3	22	100

(b)	wastewater disposal	biosolids disposal	supply chain activities	Total
Metals	2	40	53	94.4
Organics	3	0.07	2	5.4
Total	5	40	55	100

(c)	wastewater disposal	biosolids disposal	supply chain activities	Total
Metals	2	91	1.4	95
Organics	5	0.02	0.3	5
Total	7	91	1.7	100

APPENDIX B2 CASE STUDY #1 – THE DIVERSE ENVIRONMENTAL BURDEN OF CITY-SCALE URBAN WATER SYSTEMS

– SUPPLEMENTARY INFORMATION

This section contains the Supplementary Information appended to the following manuscript, which has been submitted to Water Research:

Lane, J.L., de Haas, D.W., Lant, P. (under review) The diverse environmental burden of city-scale urban water systems. *Water Research*

THE DIVERSE ENVIRONMENTAL BURDEN OF CITY-SCALE URBAN WATER SYSTEMS

- Supplementary Information -

Lane, J.L.[1], de Haas, D.W.[1,2], Lant, P.A.[1]

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S1. SCENARIO DEFINITION

These scenarios draw on previous analysis (de Haas et al. 2011, Lane et al. 2011), but have been modified to provide better definition and comparison of the water cycle associated with urban areas. A number of the key inventory items have also been updated for: (a) the infrastructure components installed since the 2008 study period; (b) those aspects where directly measured data was not available (e.g. rainwater tanks performance; fugitive greenhouse gas emissions). Finally, the analysis presented here incorporates an updated review of best practice in Life Cycle Impact Assessment models, allowing an increased focus on the challenges in utilising these models for urban water systems analysis.

The population characteristics (Table s1) and residential water use profile (Table s2) for both scenarios are summarised below. Table s3 provides an overview of the water balance for the two scenarios.

s1.1. ‘Traditional infrastructure’ scenario

The ‘Traditional infrastructure’ scenario includes the water supply and wastewater treatment systems that were in operation in the Gold Coast district during the 2007/08 period. Analysis of this scenario assumed that the population and infrastructure mix was unchanged over this one year period.

Infrastructure configuration

The mains water supply system incorporates the two large water supply dams, and two potable water treatment plants (WTPs), that directly supply the Gold Coast area. Both WTPs feed into an integrated mains water reticulation network, supplying urban residential and non-residential customers. In 2008, this mains water network also serviced 20,000 peri-urban properties that did not have connection to centralised sewers. These households, and the distribution networks serving their locations, were excluded from this scenario.

In this scenario, it was estimated that 15% of the residences in 2008 used rainwater tanks to provide water directly to certain end uses, based on tank installation records provided by the water utility. Based on surveys collated by the state government (summarised in Gardiner 2009) , it was assumed that 70% of these had been retrofitted to existing housing stock, and were used only for external demands (e.g. gardening). The remaining 30% were assumed to be connected to toilet, laundry and garden uses, as was the norm for all new housing being constructed in the region at that time. While not explicitly mandated, the installation of household scale rainwater tanks were the normal approach used to meet government regulations (QG 2008) making it compulsory for all new housing to incorporate some onsite water supply infrastructure.

The scenario’s wastewater infrastructure incorporated the four distinct sewer networks in operation during the 2007/08 period, collecting sewage from urban residential and non-residential customers. It also included the four associated sewage treatment plants (STPs), that utilise a mix of advanced biological nutrient removal systems to produce low nutrient effluents of varying quality (2-10 mg-N/L ; 1-4 mg-P/L). On average across the four STPs, 80% of the treated wastewater is discharged to the ocean, with the remaining 20% being reticulated (without further treatment) through a separate piping network for reuse to a range of non-residential customers (predominantly for irrigation of golf courses and public open spaces). All the STP biosolids are dewatered, then trucked from the STPs for application to farm lands without any further processing.

Table s1: Population characteristics for the two scenarios, & the breakdown of water supply configurations used for each residence type

	'Traditional infrastructure'	'Diversified infrastructure'	Increase (overall)
total residences (split by water supply configuration)	*202,000*	*550,235*	*348,235 (2.7×)*
mains supply	171,894	171,894	0
mains supply + rainwater tank (external use)	22,114	22,114	0
mains supply + rainwater tank (laundry, toilet & external use)	7,992	324,341	316,349
mains supply + rainwater tank (laundry use) + recycled water (toilet & external use)	0	31,886	31,886

Water & sewage balances

To construct the scenario water balance, bulk water supply (residential and non-residential) data for the 2007/08 period were taken from the 2007/08 National Performance Report for urban water utilities (NWC 2009). From that were obtained: (a) the number of urban households (202,000) connected to both mains supply and sewage networks; and (b) data used to estimate the portion (8%) of mains supply lost as leaks in the distribution system, which were assigned equally to all mains users on a flow-weighted basis.

The household mains water use profile was based on 2008 data for the Gold Coast city (Willis et al. 2009). This overall usage rate is quite low compared with local historical data (Beal et al. in press), and that of other low-density urban areas in Australia and other developed countries (e.g. Cahill and Lund 2013, Willis et al. 2011, Willis et al. 2013). However, it is well within the range of residential

water usage in other higher density or more water-stressed urban settings (e.g. Nixon et al. 2003, Saurí 2013). Following an extended period of intensive campaigns to encourage the adoption of more water-efficient infrastructure and behaviours, it is also largely representative of the new norm in urban centres of the Gold Coast and neighbouring regions (Beal et al. in press). The exception is the extremely low usage of water for outdoor purposes, based on data in winter months when gardening demands are low (Willis et al. 2009). Residential outdoor water use rates in the region have increased substantially, as regulatory restrictions on garden water use have lifted since the end of the severe drought that afflicted the South East Queensland region at that time (Beal et al. in press).

Finalising the micro-scale water use balance required an estimate of rainwater contributions from those houses with a rainwater tank installed. No such empirical data was available, hence rainwater yields were estimated using a first principles model of rainwater tank performance (section s3.2).

Table s2 – Residential water use and sewage generation estimates used in developing the water balances for both scenarios. The estimates incorporate empirical data for the Gold Coast in the 2008 study period (Willis et al. 2009), supplemented with predictions for the effect of non-mains supplies on the residential water balance (Willis et al. 2010).

	avg individual (L/p/d)	avg household (L/hh/d)
Internal use		
toilet	*21*	55
laundry (cold water use)	30	78
other	85	222
External use		
baseline (all households)	21	54
additional amount induced by the availability of a non-mains water supply source	15	39
Leaks	2	5
Total water use		
residences with mains water as the only supply source	159	415
residences supplied with mains water, and also directly from rainwater tanks and/or recycled wastewater	174	454
Sewage generation (all residences)	137	355

The water balance was then closed, by reconciling the micro-scale mains water use data with the bulk mains supply data for 2007/08. To do this, it was assumed that all households used an equal baseline amount of water for external purposes, regardless of whether that water is supplied from the mains system or directly from an onsite rainwater tank. This baseline was calculated by adjusting the rainwater tank contributions until a balance was obtained between the mains supply, mains use, and rainwater tank flow estimates. Houses with a rainwater tank were then assumed to use additional rainwater for external purposes, over and above the baseline external usage. The amount of this additional external use was informed by the predictions generated for before and after the integration of a non-mains water supply at Gold Coast housing (Willis et al. 2010).

Household sewage generation was assumed to equal total household internal water use. From this, the total residential contribution to sewage flows was estimated. Gold Coast Water provided estimates of the total dry weather sewage flows for the 2007/08 period. Non-residential sewage generation was set as the difference between this and the residential estimate.

s1.2. ‘Diversified infrastructure’ scenario

The ‘Diversified infrastructure’ scenario incorporates the full infrastructure set of the ‘Traditional infrastructure’ scenario, supplementing this with the additional infrastructure (water supply and wastewater) required to service an increased population base. While this ‘diversified’ infrastructure mix is a hypothetical one, the technology choices and infrastructure scale are based on systems that have been implemented at the Gold Coast, or in neighbouring regions, since 2008.

As with the ‘traditional’ alternative, analysis of this scenario assumed that the population, infrastructure mix, and infrastructure supply rate were unchanged over a one year period. No consideration was given to incremental trajectories for infrastructure development.

This scenario was designed in a four-step process, solving across a number of unknown variables. The hypothetical water supply infrastructure mix provided the starting point (step 1). An unchanged

estimate for average per-capita water use was adopted (step 2), so as to constrain any differences between the two scenarios to those associated with the change in water supply technology mix. Requirements for mains water distribution (step 3) and sewage management (step 4) were defined.

Table s3: Water and Wastewater balances for each scenario (in ML/d). The overall increases in water supply and sewage flows are in line with the 2.7× increase in population for the 'Diversified infrastructure' scenario. The slightly higher increase (2.8×) in the total water supply rate results from the additional demand increase 'induced' by the large-scale presence of non-mains water supply technologies in the 'Diversified infrastructure' scenario.

	'Traditional infrastructure'	'Diversified infrastructure'	Increase (overall)		Increase (per HH)
Water use - total	152	422	*270*	*(2.8x)*	+2%
residential	85	241	156	*(2.8x)*	+4%
non-residential	56	155	99	*(2.7x)*[a]	0%[a]
distribution losses	11	26	15	*(2.4x)*	-12%
Water supply - total	152	422	*270*	*(2.8x)*	+2%
dam supplies - environmental	128	131	3	*(1.0x)*	-62%
dam supplies - IPR	--	45	45		
seawater desalination	--	125	125		
rain water	3	59	56	*(18x)*	+574%
direct wastewater reuse	21	62	41	*(2.9x)*	+7%
Sewage generation - total	108	296	*188*	*(2.7x)*	0%
sewage - residential	73	199	126	*(2.7x)*	0%
sewage - non residential	35	97	62	*(2.7x)*[a]	0%[a]
Wastewater disposal - total	108	296	*188*	*(2.7x)*	0%
wastewater to reuse	21	107	86	*(5.1x)*	+86%
wastewater to sea	87	189	102	*(2.2x)*	-20%
Wastewater to reuse - total	21	107	*86*	*(5.1x)*	+86%
Class B reuse - non residential	21	55	34	*(2.6x)*	-5%
Class A+ reuse - non residential	--	3	3		
Class A+ reuse - residential	--	4	4		
IPR product water	--	45	45		
Wastewater to sea – total [b]	87	189	*102*	*(2.2x)*	-20%
secondary effluent to sea	87	170	83	*(2.0x)*	-28%
tertiary effluent to sea	--	10	10		
AWTP RO brine to sea	--	9	9		

[a] *both scenarios used the same per-household sewage flows and non-residential water demands. The apparent increases in per-HH flows shown in the results presented here are due to rounding errors in collation of the results.*

[b] *excludes brine from seawater desalination*

Step 1. Define the water supply infrastructure

A seawater reverse osmosis (RO) desalination plant was included, operating at the full capacity (125 ML/d) of the Gold Coast facility commissioned in 2009 (Poussade et al. 2011). Potable quality product water from this plant is discharged into existing mains water supply reservoirs.

Total supply from the existing dams was increased by 38%, through a combination of two separate changes. The majority (45 ML/d) of these additional dam extractions are made possible by incorporating an Indirect Potable Reuse (IPR) system that treats secondary treated effluent from a local STP, discharging its product water into the existing Hinze Dam. This IPR system is a hypothetical one, based on an existing facility in a neighbouring region (see below), and reflecting an option previously considered for implementation at the Gold Coast (CH2MHill 2008). A smaller increase (3 ML/d) was attributed to the raising of the Hinze Dam wall that occurred in 2011. This change is the difference between the 2008 yield, and the long term average yield limit specified in

regional planning studies (QWC 2010). WTP throughput was increased in proportion to the increase in dam supply rates.

A direct (non-potable) wastewater reuse system was included, based on the infrastructure commissioned in 2009 for the Pimpama-Coomera sub-region of the Gold Coast. All treated secondary effluent is processed by the AWTP, with the throughput (17 ML/d sewage flow) matched to the capacity of the infrastructure in place at that time. Product water is directed to households (for toilet and external uses) and non-residential customers via a dedicated distribution network, with any surplus product water being discharged to the sea.

A rainwater tank was included with each additional household incorporated into the 'Diversified infrastructure' scenario, consistent with policies in place, and common practice, at the time. The regulations governing new detached housing specified the need for a 5000L rainwater tank supplying all laundry (cold water only), toilet and outdoor uses. The exception was for those houses also connected to a Class A+ recycled water supply, in which case the rainwater tank was assumed to supply only the cold water laundry demands. All the rainwater tanks of the 'Traditional infrastructure' scenario were included, without any allowance for further tanks being retrofitted into the 2008 housing stock.

Step 2. Define the demand profile and number of households

In order to assess the implications of changes in water supply technology, rather than population growth, both scenarios used the same per capita 'baseline' water demand profile (Table s2). To accommodate the increased water supply capacity inherent in the 'Diversified Infrastructure' scenario, the population base and total water demand were also increased substantially (see Table s1 and Table s3).

Non-residential water use was increased in proportion to the overall increase in household numbers for the 'Diversified Infrastructure' scenario. It was also assumed that the fraction of treated wastewater diverted for direct non-residential use would remain at 20%. This was met primarily by an increase (of 34 ML/d) in the quantity of Class B recycling, although a small portion (3 ML/d) was serviced by the recycling of higher grade A+ water. Mains water supplies were used to satisfy the remainder of the increased non-residential demand.

Residential mains water demand was calculated net of the yields from rainwater tanks (59 ML/d) and the Class A+ recycling system (4 ML/d). The availability of one (or both) of these non-mains supplies was assumed to induce an additional amount (39 L/hh/d) of external water use over and beyond the baseline, consistent with the assumption for induced demand increase used in the 'Traditional infrastructure' scenario.

Mains network loss rates (8%) were unchanged from the 'Traditional infrastructure' scenario, applied to both the dam based and desalination mains water sources.

The number of additional households (~348,000) for inclusion in the 'Diversified infrastructure' scenario was then calculated, such that a balance was obtained between the overall mains water supply and demand. This represents a 2.7× increase in population, which is not inconsistent with urban growth forecasts for this region (QWC 2010). The slightly larger increase (2.8×) in overall water use is explained by the additional demand induced by the presence of so many household scale rainwater tanks, which make an important contribution (15% of total water use) to the overall supply balance of the 'Diversified Infrastructure' scenario.

Step 3. The change in urban footprint

As described above, it was assumed that all the additional residences would be accommodated by an expansion in the urban footprint. Of these, 250,000 were assumed to fall within the existing Gold Coast region, based on forecasts for local urban growth over a 50 year period (DIP 2008). The remainder (98,000) were allocated to the neighbouring region to the north of the Gold Coast. For these, additional bulk water pumping infrastructure was included in order to model the transfer of water beyond the natural Gold Coast catchment area.

Step 4. Changes in the sewage management system

It was assumed that all the additional households of the 'Diversified infrastructure' 'scenario would be connected to the sewer, with the same per-household sewage generation rate was adopted as for the 'Traditional infrastructure' scenario. Non-residential sewage generation was increased in proportion to the total number of households, and distributed evenly across all STP catchments.

A portion (~32,000) of the new households were assigned to the Pimpama-Coomera STP, thereby indirectly supplying the advanced wastewater treatment plant and non-potable reuse system.

All other sewage management under the 'Diversified infrastructure' scenario was more conventional in nature. The basis for this was a weighted average of the physical infrastructure and operations required for the four existing STPs included in the 'Traditional infrastructure' scenario. These inventories were scaled up in proportion to the increased number of households included in this scenario. As per current practice, we assumed that 100% of STP biosolids would go to agricultural reuse applications.

S2. INFRASTRUCTURE STOCK & CONSTRUCTION INVENTORIES

s2.1. Component overviews

Dams

Hinze Dam and Little Nerang Dam are the two major freshwater supply sources within the Gold Coast region. Construction inventories for these dams comprised the materials used in each of the dam walls, taken from information published by the Hinze Dam Alliance (2007) and SEQ Water (2009a, b).

For the 'Diversified infrastructure' scenario, this also included the additional materials required for the raising of the Hinze Dam wall that occurred in 2011. While this doubled the total storage volume, the majority of this increase was intended to provide a buffer for flood mitigation. Furthermore, catchment planning studies around the same time revised down the expected sustainable yields from Hinze Dam, meaning the dam wall raising only increased the dam yield for our scenario by 3%.

Water Treatment Plants

The Gold Coast region has two mains water treatment plants (WTPs) interconnected with the Hinze and Little Nerang Dams. Both plants use conventional treatment processes (flocculation, filtration, chlorine disinfection) to produce potable quality water, which is discharged into an interconnected grid that spans the entire Gold Coast city.

WTP construction inventories were based on data from the Ecoinvent database (Frischknecht et al. 2007), scaled to the capacity of the local plants[1].

No expansion of this infrastructure was required for the 'Diversified infrastructure' scenario, as that available capacity is sufficient to meet the required throughput of both scenarios.

Seawater Desalination

[1] Treatment Plant construction inventories were scaled from other data sources in proportion to their capacity, using the following approach:

$$I_{TP} = I_{DS} \times \left(\frac{C_{TP}}{C_{DS}}\right)^{0.65}$$

where: I_{TP} = quantity of required inventory item (I) for the treatment plant model used here
I_{DS} = quantity of required inventory item (I) in the datasource
C_{TP} = volumetric capacity (C) of the treatment plant in our scenario
C_{DS} = volumetric capacity (C) of the treatment plant modelled in the datasource

The Tugun seawater desalination plant is located in the Gold Coast city, using three MF-RO trains to produce treated water that is then discharged into the local mains supply network.

Construction estimates for the desalination plant and RO concentrate disposal pipeline were scaled up from the data of Muñoz et al (2008)[1].

For the product water delivery pipeline, information on the pipeline configuration (GCD 2006) was combined with the construction inventories of Grant and Opray (2005) – see also Sharma et al. (2009).

Rainwater Tanks

As described in Section s1, the two infrastructure scenarios incorporated a mix of rainwater tanks that were either (a) retrofitted to housing stock existing prior to 2007; or (b) installed with each new residence constructed after planning regulations were changed at that time.

For the latter category, those planning regulations specified the minimum rainwater tank volume required for newly constructed housing at the Gold Coast as 5000L for detached residences, and 3000L in smaller, semi-detached residences. Detached dwellings are the dominant housing type in the Gold Coast and surrounding urban areas. While the prevalence of smaller, higher density housing might be expected to grow substantially with urban infill development, the available installation data also indicates that many residences have installed rainwater tanks much larger than the minimum specified size. In the absence of any better information, we assumed an average size of 5000L, across all those rainwater tanks installed under this category.

In the years immediately prior to and following 2008, the majority of rainwater tanks retrofitted to existing housing stock were subsidised by government rebate programs. Rebates were generally only provided for tanks 5000L or greater in size. We assumed that all rainwater tanks falling under the 'retrofit' category were also 5000L in size.

Anecdotal evidence suggests that polyethylene was the preferred material for rainwater tanks installed in the Gold Coast and surrounding regions, over the time of the study period. We assumed that 100% of the rainwater tanks in this study were constructed in polyethylene, with the tank installation inventories based on work in a previous Australian study (Hallman et al. 2003).

Distribution networks for mains water supply, sewage collection, & class B wastewater recycling

The water utility managing the Gold Coast infrastructure provided detailed data on the piping size, length, and materials used for the water supply and sewage distribution systems in place during 2008, and for the bulk recycling of treated wastewater (Table s4).

Manufacturing inventories were taken from the Life Cycle Strategies database (Grant 2012), otherwise from Ecoinvent data (Frischknecht et al. 2007). In either case, the majority of the manufacturing inventories are based on Ecoinvent v2 data, customised to use Australian electricity and other inputs. Assumed transport distances (for delivery to the Gold Coast city) are also provided in Table s10.

Locally specific inventories have been compiled for certain Australian manufactured chemicals (Alvarez-Gaitan et al. 2013), however the publicly available versions of that data are presented in the units of midpoint LCIA indicators, using different impact assessment models than those chosen for this study. It was therefore not possible to incorporate that data into the models used here.

The Alvarez-Gaitan et al. (2013) study highlights that product inventories based only on process databases (such as those used here) can substantially underestimate the full supply chain impacts of certain chemicals used by the water industry. For example, incorporating the more complete supply chain coverage that is possible when input-output based databases are used, increases the GHG footprint of Alum Sulphate (one of the higher volume chemicals used in this study) by 60%.

The importance of this methodological choice, and in using locally relevant production data, could be critical to LCA studies focussed on specific water cycle components (e.g. mains water treatment) where chemicals use is a major operational input. However, the effect on our study would be far

less, since chemicals manufacturing makes a relatively small contribution to the overall LCIA results (e.g. <7% of the total GHG footprint). While improving the quality of the chemicals manufacturing inventories would be advantageous for urban water managers wishing to understand their life cycle environmental burdens, it would not change the generalised conclusions drawn in this study.

Table s4: Pipeline network lengths (km) used for the 'Traditional Infrastructure' scenario

Pipe diameter (mm)		0050	0075	0100	0150	0225	0300	0325	0375	0450	0525	0600	0750	0975	1200
Mains	DICL					201	69		35	33					
	MSCL											73	42		
	RCP					1,308	49			57		38			
	UPVC			616	461	147	41								
Sewer	RCP					122	49			31					
	UPVC				2,140	91	55								
	VIC					98									
Class B	DICL					5	5		3	4	9				
	HDPE	1	2			1		2							
	MSCL								2			2	6	6	15
	UPVC			3	6	17	9								

DICL = cement-lined ductile iron; MSCL = cement-lined mild steel; RCP = reinforced concrete pipe; UPVC = unplasticised PVC; VIC = vitrified clay; HDPE = high density polyethylene;

Construction inventories for the pipe laying were based on the work of Grant and Opray (2005) – see also Sharma et al. (2009). Section s2.2 provides more details on the pipeline construction inventories.

For the intermediate reservoirs spread throughout the water supply network, materials inventories were estimated based on data provided by the Queensland Water Commission (QWC 2009).

For the 'Diversified infrastructure' scenario, the physical infrastructure stock of these piping networks were increased in proportion to the number of additional residences. In part this reflects the expectation that much of future population growth in Australia's cities will happen by expansion of the urban footprint, rather than by urban densification. It is also an assumption that is deliberately conservative on the high side, so as to explore the potential significance of the piping infrastructure under the more complex and energy intensive water supply mix.

Sewage treatment plants

In 2008, the Gold Coast region had 4 major sewage treatment plants, each connected to their own distinct sewer system. Construction inventories for these four STPs, in the 'Traditional infrastructure' scenario, were scaled from those provided in the Ecoinvent v2 database (Frischknecht et al. 2007)[1].
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The sewage treatment infrastructure in the 'Diversified infrastructure' scenario is a scaled up representation of those same four plants, with the only exception being the inclusion of the additional Pimpama-Coomera STP as described below. The plant construction inventories were scaled up linearly in proportion to the flow increase, assuming that this population increase would be serviced by the construction of new plants, rather than by expanding the capacity of the existing treatment plants.

Pimpama-Coomera secondary treatment, advanced treatment, and wastewater recycling for urban use

The Pimpama-Coomera sewage treatment plant (STP) and advanced wastewater treatment plant (AWTP) were commissioned at the Gold Coast in 2009. The throughput of both plants was based on the expected dry-weather capacity (17.1 ML/d) of the STP infrastructure installed at that time. Construction inventories for the STP and AWTP were based on literature information (Friedrich 2001, Frischknecht et al. 2007)[1].

Table s5: Pipeline network lengths (km) used for the Class A+ reticulation system of the 'Diversified Infrastructure' scenario

Pipe diameter (mm)	0100	0150	0225	0300	0315	0525	0600	0750	0975
DICL				15	7	26	15		
HDPE	296			10					
MSCL								23	18
UPVC		244	108	31	12				

DICL = cement-lined ductile iron; MSCL = cement-lined mild steel; UPVC = unplasticised PVC; HDPE = high density polyethylene;

AWTP product water is distributed through a reticulation network for reuse in nearby residential and non-residential properties. Detailed pipe length and materials data for this network was provided by GCW, and scaled up to match the number of households included in the Pimpama-Coomera reuse scenario for this study (Table s5). Construction inventories for the piping system were developed as for the other Gold Coast reticulation networks described above.

Advanced wastewater treatment for indirect potable reuse

The AWTP for the hypothetical Indirect Potable Reuse (IPR) system was sized at the 2007/08 capacity (54 ML/d) of the Coombabah STP, and based on the Bundamba AWTP of the Western Corridor scheme in the Brisbane River catchment (O'Toole et al. 2008, Traves et al. 2008). An estimate of plant construction inventories was generated by scaling up the data provided by Muñoz and Fernandez-Alba (2008)[1].

The length, diameter and materials selection for the product water pipeline (from the Coombabah STP to Hinze Dam) was estimated based on information in CH2MHill (2008). Pipeline construction inventories were estimated using a similar approach to those for the reticulation systems described above. It was assumed that discharge of the RO concentrate would utilise the existing secondary effluent disposal pipeline from the Coombabah STP.

s2.2. Pipeline network construction inventories

All major and network pipelines were assumed to be installed underground. The installation inventories are based on data compiled for a study by Grant and Opray (2005), later reported by Sharma et al. (2009).

An example of the construction inventory is provided for a 225mm DICL pipe in Table s6.

s2.3. Infrastructure lifespans

Annualised construction inventories were estimated by dividing the overall inventory by the nominated lifespan for the infrastructure item. Treatment plant inventories were annualised based on the lifespan assumptions used in the source dataset. Other inventory items were annualised using lifespans listed in Table s7.

Table s6: Inventory estimates for the installation of 225mm DICL pipe

Pipe manufacture	ductile Iron (desulphurised) =	39.3	kg/m of pipe
	cement lining for MSCL & DICL pipes =	12.8	kg/m of pipe
	electricity use for casting =	2.3	kWh/kg of pipe
Pipe supply	transport distance =	100	km
	transport requirement =	5.2	t.km/m of pipe
Gravel supply	gravel used in the trench =	766	kg/m of trench
	gravel waste at processing site =	0.08	kg-wasted/kg-used
	electricity use for gravel washing =	0.005	kWh/kg of gravel product
	transport distance =	40	km
	transport requirement to supply gravel =	31	t.km/m of trench
Excavate trench	operation time =	0.086	h/m of pipe
	diesel use =	0.131	kg/h
	lubricant use =	0.003	kg/h
Remove spoil	spoil quantity =	0.9	kg-spoil / kg-gravel
	transport distance =	15	km
	transport requirement to remove spoil =	10	t.km/m of trench

Table s7: Assumed lifespan for infrastructure component items

Item	Lifespan (yrs)
Underground pipes (all material types)	50
Industrial pumps	15
Household rainwater pumps	10

Item	Lifespan (yrs)
Concrete tanks	50
Polyethylene tanks	25
Steel tanks	20

S3. INFRASTRUCTURE OPERATIONS INVENTORIES

The following sections describe the approaches used to model operational performance for each of the components of the 'Traditional infrastructure' and 'Diversified infrastructure' scenarios.

Wherever possible, the inventories are based on measured data for the actual systems in place. For the Gold Coast-based infrastructure components of the 'Traditional infrastructure' scenario, detailed operational summaries were provided by the local water utility (*Gold Coast Water*). For the wastewater recycling infrastructure added with the 'Diversified infrastructure' scenario, operational data was provided by the respective owners of the Pimpama-Coomera wastewater system (*Gold Coast Water*) and the nearby Bundamba AWTP (*Water Secure*).

s3.1. Mains water supply

Dam supplies

For the pumping of raw water from the dams to water treatment plants, annual average electricity use data was provided by the water utility for the 2007-2008 period. This was scaled up linearly for the increased extraction rates of the 'Diversified infrastructure' scenario.

As described in Section s5.1, dam evaporation losses were not accounted for in this study.

The dam inventories do include an estimate of ongoing CH_4 emissions caused by the degradation of carbon matter (vegetation, sediment) flowing into the storages. The methodology for these estimates is described in Section s3.4.

Dam water treatment

Annual 2007-08 data for operational inputs (raw water feeds; use of electricity, fuel & chemicals) and outputs (treated water discharge; sludge waste disposal) for both WTPs was provided by the water utility.

Sludge wastes from both WTPs are sent to the sewer, and their metals content was estimated from the chemical dosing levels. These metals were accounted for in the STP modelling (see Section s3.3), where it was assumed that they report entirely to the biosolids stream.

For the 'Diversified infrastructure' scenario, all flows were scaled up linearly with the increased WTP throughput. The exception was for electricity and fuel use, where a portion (30%) of the WTP inputs in the historical data were assumed to be fixed over time.

Mains network supply

The Gold Coast water utility provided annual 2007-08 data on electricity use for mains distribution pumping, and it was assumed that this would scale up linearly with the increased throughput of the 'Diversified infrastructure' scenario.

The 2007/08 operating period involved no export or import of bulk mains water from/to the Gold Coast (NWC 2009). However the 'Diversified infrastructure' scenario included ~100,000 households located in the adjoining regions to the north of the Gold Coast, and it was assumed that additional pumping was required to provide mains supplies to these. The electricity use required for this additional pumping was based on a previous estimate for the energy-intensity of bulk water transfer north from the Gold Coast (Hall et al. 2009).

Seawater desalination

Electricity use data for the seawater desalination plant was taken from (Poussade et al. 2011). Detailed inventories for other operational inputs (power, fuel, chemicals, membranes) and outputs (RO concentrate, sludge) were not available at the time this study was undertaken, so were developed by combining information from Muñoz et al (2008), Water Corporation (2008), Vince et al (2009), Mrayed & Leslie (2009) and Leslie (2010), accounting for differences in local water chemistry.

Reverse Osmosis concentrate (brine) and waste solids streams are mixed together and discharged to sea through a dedicated pipeline. The impurities in this flow were not accounted for in the impact assessment models.

Product water from the Gold Coast desalination plant is pumped through a dedicated pipeline, via an intermediate booster pump station, to three different pre-existing mains supply reservoirs along the pipeline route. For this study it was assumed that the product water would be evenly distributed to users along the length of the north-south delivery pipeline, and that only a portion (20%) of the product water would reach the intermediate transfer pump station. Estimates of energy use by two product water pump stations were based on the values provided in previous studies (Hall et al. 2011, Poussade et al. 2011). It was assumed that all supply from the reservoirs to end-users was by gravity feed. Bulk distribution losses were set equal to the assumed value (8%) for the Gold Coast mains network.

s3.2. Rainwater Tank Supply

Very little empirical data is available on the actual operational performance of the large number of rainwater tanks that have been installed in the Gold Coast and nearby cities. Our modelling of rainwater tank operations was therefore based on information derived from a number of sources.

Rainwater yields

As described in Section s1, this study adopts three different end-use configurations for rainwater tank systems. These involve rainwater being used for (i) external use only; (ii) laundry use only; and (iii) external, laundry and toilet uses.

A separate estimate for rainwater yield was generated for each of these configurations (Table s8), using a daily timestep model for rainwater tank systems in Australia (Vieritz et al. 2007) applied to 105 years of historical Gold Coast climate data. No account was made for the possibility of climate change induced shifts in future rainfall or evapotranspiration patterns. In each case, the connected roof area was set to 100m^2, being the legislated minimum (at that time) for new detached housing stock (QG 2008). The default software assumptions were used for estimates of roof losses, first flush volume and other supply side parameters.

Table s8: Household rainwater tank yields used in this study

Connected end-use	Rainwater yield (kl/hh/y)
External only	24
Laundry only	28
Toilet + Laundry + External	62

Mains water backup systems

Mains water backup requirements were calculated as the difference between the rainwater yield and the total end-use demands (from Table s2). Of the 'Traditional infrastructure' rainwater tanks plumbed solely for external use, 50% were assumed to have mains backup water provided through a manually operated bypass, with the remainder utilising a mechanical float valve to top up the tank as required. For both infrastructure scenarios, all rainwater tanks being plumbed to laundry only, or external/toilet/laundry, were assumed to use an automated switch to bypass the tank when backup water is required. Anecdotal information indicates this rapidly became the norm for new housing stock as the number of installed rainwater tanks in the years following 2008. It is assumed that electricity use for these bypass switches is incorporated into the pumping energy data described below.

Delivery systems

As per common practice in Australian cities, both scenarios assumed that all rainwater tanks use an electrical pump to deliver 100% of the tank water to the various end-uses, without any water treatment (such as UV sterilisation) being applied.

Electricity use estimates for supplying toilet and laundry end uses were based on data from Tjandraatmadja et al. (2013). That study used a controlled laboratory system to simulate Australian household conditions, generating data on rainwater distribution power demand for a variety of fixed-speed pump sizes (0.2kW, 0.55kW, 0.75kW) typical of Australian household installations, with and without the installation of a pressure vessel in the delivery system.

Anecdotal evidence from the time of this study, indicated a tendency for households to install pumps at the larger end of the available range in retail stores, with relatively low uptake of pressure vessel based systems. To this end, it was assumed that 33% of installations would use a delivery pump with a 0.55kW motor (1.46kWh/kL for supply to toilets; 1.47 kWh/kL to laundry), 33% would use a 0.75kW motor (3.25 kWh/kL to toilets; 1.70 kWh/kL to laundry), and 33% would use a 0.75kW motor coupled with a pressure tank (2.00 kWh/kL to toilets; 1.60 kWh/kL to laundry). Those estimates for supply to toilets are averaged across the data for full flush and half flush events from Tjandraatmadja et al. (2013), assuming that full flush events account for 63% of the total toilet flushing volumes in our study household. The half/full flush split was derived from Gold Coast end-use event statistics in (Beal and Stewart 2011).

Tjandraatmadja et al. (2013) do not provide equivalent data for pumping water to outdoor uses, hence estimates for power use required to deliver water to external uses was taken from Siems et al. (2013). That latter study used high resolution (5 second intervals) water and energy monitoring equipment to quantify the power draw associated with single end use events, for a variety of end use types, across 19 different dwellings. They calculated a specific power use of 1.02 kWh/kL, taken as a single average across all outdoor usage events for all households in the study. That estimate (1.02 kWh/kL) was used for all three delivery system configurations supplying to external uses. The use of a consistent value across the different delivery configurations was considered appropriate,

given: (a) outdoor usage events involve relatively high flow rates (13L/min or greater) over extended durations (Talebpour et al. 2011, Tjandraatmadja et al. 2012); (b) at such flow rates, both the large pumps considered in this study would be operating at an efficient point on their pump curves, and have a very similar specific power use (Tjandraatmadja et al. 2013); and (c) the extended duration pumping events imply that installing a pressure vessel in the delivery system would have only a minor effect on overall pumping power use (Tjandraatmadja et al. 2013).

The majority of household rainwater tank systems installed in South East Queensland (to date), also utilise a pumping control system that can draw a substantial baseload current. An estimate of 2.2W was used for all pumping system configurations. This was the average of 13 data points measured for a range of 1st and 2nd generation control system products, taken from Hauber-Davidson and Shortt (2011) and Tjandraatmadja et al. (2012), and from field data obtained during this study.

Overall, the average power use (in kWh per kL of water pumped) by the three different end use configurations (external use = 1.8 kWh/kL; laundry use = 2.3 kWh/kL; external/laundry/toilet uses = 1.85 kWh/kL) fall well within the range of field data (0.3 – 11 kWh/kL) produced in various Australian trials for rainwater delivery systems that do not include a treatment component (Cunio and Sproule 2009, Hauber-Davidson and Shortt 2011, Hood et al. 2010, Retamal et al. 2009, Retamal and Turner 2010, Siems et al. 2013, Talebpour et al. 2011, Tjandraatmadja et al. 2013, Umapathi et al. 2013, WCG 2009).

The consensus from those studies is that the overall rainwater pump energy intensity varies substantially across households in Australia, depending on a complex mix of water use (e.g. end use type & amount; resident behaviour), delivery system (e.g. pump type, size & brand; control system & pressure management systems in place; elevations) and appliance (e.g. toilet filling mechanisms; washing machine fill pattern) characteristics. A large part of this variability can be attributed to inefficiencies created by poorly aligned supply and demand characteristics of the system, such as the installation of pumps that are incorrectly sized for the required demand profile (Hauber-Davidson and Shortt 2011, Retamal et al. 2009, Tjandraatmadja et al. 2013).

The overall volume-weighted averages for rainwater tank supplies in the 'Traditional infrastructure' scenario (1.83 kWh/kL of water pumped) and 'Diversified infrastructure' scenario (1.87 kWh/kL of water pumped) are higher than the median (1.40 kWh/kL) of empirical values identified in a recent review of the available literature (Vieira et al. 2014). There is insufficient evidence in the literature to ascertain whether or not that available data is representative of the current stock of rainwater tanks in Australia, let alone those that would be installed in future urban growth areas. What is clear is that the energy footprint of future rainwater tank supplies could be extremely sensitive to trends in rainwater tank system design and use. The approach used here is deliberately conservative on the high side, so as to consider how substantial that energy footprint might become in the event of a large scale tank rollout.

s3.3. Wastewater treatment and recycling

Sewage collection & treatment – 'Traditional infrastructure' scenario

The local water utility provided operational data, for the 2007/08 period, for the four sewer and STP systems in existence at that time. This data included dry weather flows, influent and effluent quality (COD, nutrient & chlorine concentrations), overall biosolids generation, biosolids moisture content, and usage rates for electricity, fuel and chemicals. Data for metals concentrations in the treated wastewater was also provided, supplemented with equivalent estimates for organic micropollutants taken from literature (see section s3.5).

For each of the STP datasets, the portion of biologically generated biosolids was back calculated by subtracting (from the total) an estimate for the chemically produced solids. The latter was determined based on stoichiometry, and the known dosing rates for chemical phosphorus removal. The overall phosphorus content of biosolids was calculated by mass balance, given the known phosphorus content in the raw sewage and final treated effluent. The nitrogen and carbon content were calculated using conventional industry assumptions for the waste-activated sludge composition.

The compilation of inventories for disposal of the biosolids is described below.

Fugitive gas (CO_2, CH_4, N_2O, NH_3, NO_X, SOx, CO) emissions were estimated across all relevant stages of the wastewater management system. The main greenhouse gas inventories (CO_2, CH_4, CH_4, N_2O) were based on a combination of mass balance and emission factors generated through an extensive literature review (see section s3.4).

Sewage collection & treatment – 'Diversified infrastructure' scenario

As described in section s1.2, total sewage generation for the 'Diversified infrastructure' scenario was scaled up linearly with the increased population base. Total STP throughput (dry-weather flow) therefore increased from 108 ML/d to 295 ML/d.

A small portion of this increased sewage flow was allocated to the Pimpama-Coomera STP (see section s2), an additional installation in operation at the Gold Coast from 2009. In our scenario, the average dry weather throughput of the STP was set at 17.1 ML/d, which represented the expected full capacity of the STP infrastructure in place at the time.

Daily operational data for the Pimpama-Coomera STP was obtained from the water utility, covering the period from January 2009 to June 2010. STP throughput over this period (average 4.6 ML/d) was much lower than the capacity (17.1 ML/d) included used in the 'Diversified infrastructure' scenario. To generate inventories at the higher throughput, a linear, mass-balance based model was derived from the empirical data, linking the process inputs (chemicals and power use), and outputs (grits and biosolids), to key influent/effluent constraints (flow, N, P, COD). The STP inventories were then scaled up, subject to the following constraints:

- Influent sewage concentrations (TN, TP, COD) were set to equal the flow-weighted average of the four STPs included in the 'Traditional infrastructure' scenario.
- Chemicals usage in the model was modified to reflect usage efficiency improvements that were achieved for that particular plant in the period following June 2010.
- Based on previous STP analysis by the study team, the empirical power use was assumed to be 26% fixed (time-dependent), 50% load-dependent (e.g. for aeration) and 24% flow-dependent.
- Effluent nutrient and chlorine concentrations were set equal to those for the empirical data from the 2009-10 period.

An additional 170 ML/d of dry-weather flow increase was required to meet the specified conditions for the 'Diversified infrastructure' scenario. It was assumed that this increase would be predominantly met by the construction of additional STPs. We therefore modelled this by scaling up linearly all operational flows (inputs and outputs) for the four STPs included under the 'Traditional infrastructure' scenario.

Biosolids use on agricultural fields

Under both scenarios, all biosolids are transported 200km by truck for agricultural reuse, as per the water utility practice during the 2008-09 period. Inventories for machinery operation to apply the biosolids to soils were based on Foley et al (2010b).

Fertiliser offsets were included in the system boundary. For this, it was assumed that the biosolids would displace a mix of DAP (18% N; 20% P) and Urea (46% N) fertilisers, making an allowance for the relative bioavailability of the nutrients in the different products. For this we followed the approach of Foley et al. (2010a), assuming that the bioavailability of biosolids N and P is only half that of the conventional fertiliser alternatives. These can be considered indicative only, since biosolids application rates in Australia are typically controlled based on more sophisticated consideration of the biosolids-crop nitrogen balance. In the case of biosolids phosphorus, the bioavailability can be much lower or higher than the value used here, depending on how much of the biosolids P is bound to metal (Fe; Al) ions as a result of chemical phosphorus removal being employed in the STP (O'Connor et al. 2002, Pritchard et al. 2010). Complicating this issue is the fact that, in Australia at least, conventional biosolids application rates will often substantially exceed the crop P requirements

(Barry and Bell 2006, Pritchard et al. 2007). Locally specific biosolids-soil-crop nutrient balances would be an important requirement for LCA analysis focussed more specifically on biosolids reuse systems.

While there are some local studies indicating a risk of nutrients loss to waterways following biosolids application (Barry and Bell 2006, Pu et al. 2008), there is little information available to guide quantitative estimates of such fluxes. The ReCiPe Life Cycle Impact Assessment method (Goedkoop et al. 2009) provides default delivery ratios of N and P (applied to soils) to waterways, both for synthetic and organic fertilisers. These factors were used to quantify the net nutrient fluxes (to waterways) associated with the biosolids application. However, it should be noted that the ReCiPe assumptions are taken from modelling of European conditions, and are likely to be too high in many Australian and other regions. Our results therefore provide a potentially conservative approach, for the sake of illustrating the implications of choosing default LCA methods in the analysis of urban wastewater systems.

Secondary effluent irrigation

From the operations data provided by the Gold Coast water utility, 20% of the secondary treated effluent in 2008 was reused for a variety of purposes, primarily to irrigate golf courses and residential parkland. It was assumed that the default source of irrigation (if available at all) for these sites would be water pumped directly from local streams, and that the demand for these local supplies would outstrip the allowed extraction volumes. The availability of the Class B source could therefore increase the total irrigation volumes applied at these sites. In the absence of any supporting information to generate quantitative estimates, we assumed that the creek water offsets were only 1/3rd of the total irrigated flow.

This 20% secondary effluent reuse ratio was also applied to the full STP throughput of the 'Diversified infrastructure' scenario, net of the effluent volumes distributed to advanced treatment facilities for further processing (see below). Given the tight constraints imposed on maintaining existing environmental flow regimes in the Gold Coast and surrounding areas (QG 2009), we also assumed that no additional distributed stream extraction would be possible as the urban population increases. The additional Class B irrigation volumes under the 'Diversified infrastructure' scenario were therefore not credited with offsetting any direct extraction from local streams.

We assumed that the low nutrient concentrations in the secondary effluent would limit the opportunity for Class B irrigation to offset the use of conventional fertilisers, and constrained these offsets to match 50% of the applied N and P.

Advanced wastewater treatment for direct, non-potable reuse (DNR)

100% of the secondary effluent from the Pimpama-Coomera STP is processed further in an advanced wastewater treatment (AWTP) facility. Daily AWTP operational data was provided by the water utility, for the same operating period (January 2009 to June 2010) as that for the Pimpama-Coomera STP. As with the STP, inventories were scaled up from the empirical data to match the chosen throughput of the 'Diversified infrastructure' scenario. The following assumptions were used to do this:

- Power use was assumed to be predominantly (90%) flow-dependent because of the membrane treatment system and large product water distribution pumps involved. The remainder (10%) of power use was set as fixed (time-dependent).
- Chemical requirements for treatment increased linearly with the increased throughput.
- Effluent nutrient and chlorine concentrations were set equal to those for the 2009/10 operations.
- AWTP sludge flows were estimated based on the modelled chemicals usage, and assumed to be mixed with the STP biosolids prior to reuse on agricultural crops (as per current practice).

The assumptions relating to wastewater micropollutants (organic and metals) are described in section s3.5.

The Pimpama-Coomera AWTP product water is distributed for direct use by both residential and non-residential customers. The water balance for our scenario modelling of this reuse is described in Section s1.2. As per standard practice during the 2009-2010 period, any unused product water was assumed to be discharged to the sea.

Recycling wastewater to urban consumers also in the Pimpama-Coomera sewage catchment introduces the potential for accumulation of wastewater contaminants in the system. While the wastewater directed to external purposes (e.g. gardening) was assumed to be 'purged' from the system, the contaminants in recycled water directed to the toilets will present again to the sewage system. To maintain the use of simplified linear models for our study, we assumed that any recycled contaminants are immediately and fully removed in their next pass through the STP. We achieved this by modelling the following assumptions:

- the nitrogen content is fully nitrified and denitrified in the STP, increasing the amount of N_2 and N_2O produced;
- the phosphorous content is fully removed by chemical precipitation, with an associated increase in chemicals usage; and
- the organic micropollutants are fully oxidised, and the metals report entirely to the STP biosolids.

Advanced wastewater treatment for indirect potable reuse (IPR)

Detailed operational data was obtained for the Bundamba AWTP from the facility operator, and from published data (Poussade et al. 2011). This included 12 month averages for inputs (energy, fuels, chemicals, membranes), outputs (sludge, wastewater flows, product water) and water quality (nutrients, chlorine) for all key streams.

In order to scale the Bundamba AWTP operations to match the 'Diversified infrastructure' scenario conditions, a linear, mass-balance based model was derived from the empirical data. This model linked process inputs (chemicals and power use) and outputs (product water, sludge) to key influent (flow and nutrients) and output (nutrients) constraints.

That operations model was then used to generate inventories for the scenario AWTP according to the following:

- influent flow (54ML/d) and quality parameters (3mg-N/L; 4.3mg-P/L) were set to those of the Coombabah STP secondary effluent;
- the nutrient and chlorine concentrations in the product water and RO concentrate would remain the same as for the Bundamba AWTP;
- denitrification N2O emissions were assumed to be at the lower end of the data range measured by Foley et al (2007).

The assumptions relating to wastewater micropollutants (organic and metals) are described in section s3.5.

A power estimate for pumping the product water to Hinze Dam was based on data collected in a previous study (Hall et al. 2009, Hall et al. 2011). Losses of product water from dam evaporation or spillage were set to zero. Power use for discharging the RO concentrate was set equal to the measured data for disposal of Coombabah secondary treated effluent.

s3.4. Direct greenhouse gas emissions from urban water system infrastructure

This section summarises the rationale used to estimate direct emissions of greenhouse gases from system operations. Preference was given to locally relevant data where possible. Future analysis

should also consider other emerging science – for example, the generation of CH_4 (Chaosakul et al. 2014) and N_2O (Short et al. 2014) in gravity sewers are not covered in this review.

Table s9: summary of approaches used in this study to estimate direct gaseous emission fluxes

	source		estimation approach	
carbon (C) balance	non-biogenic carbon	- in sewage	10	% of total sewage carbon
	carbon sequestration	- biosolids to agriculture	24	% of carbon applied to soils
	CO_2 from STP & biosolids	CO_2 from STP & biosolids		*calculated by mass balance*
CH_4 generation	in sewers	in sewers	5	mg-CH_4/L of sewer flow
	in STPs	- aerobic processes	*0.005*	*methane correction factor*
		- anaerobic processes		*modelled*
	from biosolids disposal	- to agriculture	2.8	g-CH_4 per kg-ds applied
	in dams	- Hinze Dam	44	mg-CH4/d, per m^2 of surface area
		- Little Nerang Dam	261	mg-CH4/d, per m^2 of surface area
N2O generation	in STP treatment	in STP treatment	7.5	g-N_2O per kg-TN in influent
	from wastewater discharge	- to ocean	3	g-N_2O per kg-N discharged
		- irrigation	12.6	g-N_2O per kg-N discharged
	fertilisation of agricultural fields	- biosolids	15.7	g-N2O per kg-N applied
		- conventional	15.7	g-N2O per kg-N applied

Biogenic vs non-biogenic carbon

The conventional approach for water industry analysis is to assume that all sewage carbon is biogenic in origin, meaning that downstream mineralisation of this carbon is ignored in greenhouse gas accounting protocols (IPCC 2006).

However four recent studies have identified notable levels of non-biogenic (e.g. fossil) carbon in urban wastewater systems (Griffith et al. 2009, Law et al. 2013, Nara et al. 2010, Yoshida et al. 2014). The origin of this non-biogenic is thought to be surfactants and other chemical products (Griffith et al. 2009, Law et al. 2013), with substantial contributions attributed to industrial wastewater discharges into the sewer (Law et al. 2013). Mineralisation of any such carbon in the wastewater treatment process should be included in life cycle greenhouse gas footprints.

For this study, 10% of all sewage carbon is assumed to be of non-biogenic origin. That estimate is the load-weighted average of measured data for four sewer systems in regions neighbouring the Gold Coast city (Law et al. 2013). For each STP in our scenarios, an overall ratio of biogenic : non-biogenic was calculated across all the carbon inputs, accounting for the origin (biogenic vs. non-biogenic) of any chemically dosed carbon sources into the STP. In this particular case study, only a very small amount of chemical carbon dosing (for nutrient removal) is employed, using plant-derived bioethanol (100% biogenic carbon). The overall input ratio of non-biogenic carbon therefore remained at ~10% of total. That overall ratio (10% of total carbon is non-biogenic) was then applied uniformly to all carbon outflows (as CO_2; as CH_4; in the wastewater; in the biosolids) from that plant.

To our knowledge, the locally relevant data from Law et al. (2013) are the most comprehensive such values reported in the literature. Their range (4-14% of carbon is non-biogenic) agrees well with the single data point (14% of carbon in STP influent is non-biogenic) collected by Yoshida et al. (2014). Further field studies would be required to ascertain whether the assumptions used here are representative of other regions in the world.

Much higher fractions of non-biogenic carbon (25% of dissolved organic carbon; 14% of particulate organic carbon) were detected in another study focussed on treated effluent from activated sludge plants (Griffith et al. 2009), and occasionally used for the sensitivity analysis in urban water system studies (Carballa et al. 2011, de Haas et al. 2014, Rodriguez-Garcia et al. 2011). However, the utility of that data for quantifying overall non-biogenic carbon loads is questionable. Typically, residual carbon levels in such effluent streams are very low – in our study, the carbon in treated wastewater

constitutes only ~5% of the total sewage carbon load. Sewage based data points (as used here) are therefore more likely to be useful for understanding the source of carbon in the wastewater system.

Carbon sequestration

The long-term sequestration of biogenic-sourced carbon would represent a net removal of CO_2 from the atmosphere, and therefore should be given a credit in life cycle greenhouse gas analysis. In contrast, the sequestration of fossil (non-biogenic) carbon should be excluded, since this represents no net change to the concentration of radiative forcing agents in the atmosphere.

For this study, the sequestration potential is taken from the default approach (25 kg-CO_2e / kg-dry solids applied to soils) provided for use by the Canadian water industry (Brown et al. 2010). This translates to approximately 24% of the biosolids carbon being retained in the soil over the long term. To quantify the overall implications for the greenhouse gas footprint, that carbon sequestration estimate is combined with the assumptions used on the biogenic vs. non-biogenic carbon split in biosolids (see above).

The results from this approach should be interpreted with caution, since there are multiple reasons why this sequestration factor might lack relevance for many parts of the world. Firstly, the sequestration potential will vary greatly across the disparate range of other biosolids disposal methods (e.g. landfill, mine site rehabilitation, incineration) adopted around the world. For the agricultural use option considered in our case studies, soil carbon stability is likely to be dependent on climate, agricultural management practices, and a host of other mitigating factors (Baldock et al. 2012, Luo et al. 2010, Sanderman and Baldock 2010, Thorburn et al. 2013). Specifically for the Australian context, carbon sequestration from the direct addition of organic matter to local soils might well be lower than in equivalent overseas studies, given that the warmer Australian climates will encourage higher soil degradation rates (Sanderman and Baldock 2010, Sanderman et al. 2010). Furthermore, the quality of predictive soil carbon estimates will be constrained by challenges in interpreting the limited availability of longitudinal studies indicating the extent to which soil-sequestration can be maintained over the long term (Sanderman and Baldock 2010).

Given these concerns, we assume that the simplified estimate used here is at the upper bound of sequestration possible from biosolids application to Australian soils. It is not possible to determine a more realistic set of assumptions, hence any analysis focussed on biosolids carbon sequestration will involve a high degree of inherent uncertainty.

The potential for long term sequestration of carbon in irrigated wastewaters was not considered. This omission is not expected to have a significant bearing on our greenhouse-gas analysis, given the treated wastewater represents only a very small portion of the overall carbon budget (see above).

CH_4 generation in sewers

Substantial CH_4 generation has been observed in sewer networks by a number of recent Gold Coast studies (Foley et al. 2009, Guisasola et al. 2008, Guisasola et al. 2009), and in the raw sewage entering STPs in other Australian and international locations (Daelman et al. 2012, Law et al. 2012a, Wang et al. 2011).

Sewer CH_4 generation rates would be expected to vary substantially across different systems and locations. The main cause of sewer generated CH_4 is thought to be anaerobic bacterial activity occurring in rising mains, with the generation rate being sensitive to (amongst other things) residence time and temperature (Foley et al. 2009, Guisasola et al. 2009, Liu et al. in prep). The presence of trade waste inputs with highly biodegradable carbon could increase the level of CH_4 production (Sudarjanto et al. 2011), whereas certain chemicals commonly dosed to control corrosion have been shown to inhibit CH_4 production (Jiang et al. 2013, Zhang et al. 2009). While there is also some evidence of CH_4 generation in open gravity systems (e.g. Foley et al. 2009), this phenomenon is less well understood and characterised in the literature.

Given the prevalence of pressurised (rising main) sewers in the Gold Coast sewer network, sewer generation of CH_4 in both scenarios was set to equal 5 mg-CH_4 per litre of total dry weather flow.

While a somewhat arbitrary choice, this 'emission factor' represents a conservative mid-range estimate when applied across the entire sewer system for this case study. 5 mg-CH_4/L is at the lower end of measured values from the distinct sampling campaigns conducted on Gold Coast rising mains (Foley et al. 2009, Guisasola et al. 2008, Liu et al. in prep), with dissolved CH_4 concentrations as high as 25 mg-CH_4/L being detected. It is also substantially lower than the prediction of ~12mg/L by local modelling work (Foley et al. 2009, Guisasola et al. 2009), considered to be realistic for large scale pressurised sewer systems in Australia (Foley et al. 2009).

The final component of this calculation was to assume that 100% of the sewer generated CH_4 is stripped to atmosphere. No data or models were found in the literature that could support a more precise assumption. In one study, the activated sludge process oxidised up to 80% of the dissolved CH_4 in the plant feedstock, thereby avoiding a (potentially) substantial emission burden (Daelman et al. 2012). In contrast, results for one particular Australian STP indicate a much higher fraction of dissolved CH_4 in the raw sewage being stripped during the STP operations (Law et al. 2012a). Furthermore, both those studies are limited to analysis of the dissolved CH_4 reaching the STP, but provide no indication of how much sewer generated CH_4 might already have been stripped at release points (e.g. pump stations) throughout the sewer network.

Despite the availability of locally relevant empirical data, there remains a substantial degree of uncertainty associated with the estimation approach used here. The available evidence does suggest that the sewer CH_4 estimates presented here are conservative for this particular (Gold Coast) urban water system. However, this may or may not be the case for other sewer systems with very different network characteristics.

CH_4 and CO_2 generation in sewage treatment plants

None of the Gold Coast sewage treatment plants include anaerobic primary treatment steps, and only one utilises anaerobic solids digestion (with methane recovery for power generation). For the latter plant, we used a mass balance based methodology for estimating CH_4 losses that was developed by de Haas *et al.* (2009).

While CH_4 generation in managed aerobic treatment steps is normally overlooked in wastewater system greenhouse gas accounting, IPCC guidelines do note the possibility that CH_4 generation could be as high as 10% of COD removed from the wastewater. To account for the possibility that small amounts of CH_4 can be generated from aerobic treatment trains, we used the methane correction factor (0.005) suggested by de Haas et al. (2014).

For all treatment plant types, CO_2 emissions were estimated by mass balance, accounting for inflows (sewage; chemicals) and outflows (wastewater; biosolids; CH_4).

CH_4 and CO_2 generation from biosolids disposal

For biosolids disposal, we assumed a methane generation rate of 2.8 g-CH_4 per kg of dry solids applied to agricultural soils. This was taken from the median value of the literature review by Foley *et al.* (Foley and Lant 2007).

We did not account for the possibility that biosolids might be stockpiled at farms (prior to application to the soil) and therefore generate substantial quantities of methane through anaerobic degradation. The methodology proposed by Brown et al. (2010) indicates that this could have notable greenhouse gas implications, however we did not review the prevalence of this practice in Australia.

CO_2 emissions were calculated by mass balance. The biosolids carbon content was estimated based on general industry data, providing the mass of carbon applied to soils. CO_2 emissions were then calculated by mass balance, net of the assumed values for carbon sequestered in soils (see above) and the CH_4 losses.

CH_4 generation in water supply dams

While dam methane (CH_4) generation has been extensively studied in a small number of water supplies around the world (Barros et al. 2011, Bastviken et al. 2011), there is very little information

available to guide a more generalised assessment on the significance of CH_4 fluxes from specific urban water supply dams.

The Gold Coast is somewhat of an anomaly in this regard, where two recent studies have collected data from the Hinze (Sherman et al. 2012) and Little Nerang (Grinham et al. 2011, Sherman et al. 2012) dams. The available evidence indicates that these emissions are predominantly caused by ongoing carbon inputs (e.g. leaf litter) to the water bodies, rather than from the vegetation originally inundated at the time the dam was first constructed.

A methane emission estimate for the Little Nerang Dam (268 mg-CH_4/m^2/d) was set to equal the average of the three daily avg flux rates from Grinham ***et al.*** (2011), and a separate estimate from Sherman et al. (2012). The equivalent emission factor for Hinze Dam (44 mg-CH_4/m^2/d) is taken from Sherman et al. (2012). Sherman et al. (2012) also reported a range of other area-weighted average values for both dams, using different methodological approaches to deal with sampling bias. Only the higher end estimates were used here, on the premise their sampling approach may have given undue weighting to measured diffusive fluxes, and underestimated ebullition (bubble) fluxes from these storages. The Grinham ***et al.*** (2011) study, and other local data (Grinham 2012), indicate that intermittent bubbling rates can be substantial in water bodies with forested catchments such as those at the Gold Coast. Of the two dams, the Little Nerang catchment has a much higher degree of vegetation cover, presumed to be the reason for its much higher specific CH_4 flux rates.

To calculate the overall (average annual) CH_4 flux, those emission factors were combined with estimates for water body surface area at the nominal Full Supply Level. For Little Nerang Dam, a surface area of 49ha was used in both scenarios. For the 'Traditional Infrastructure' scenario, the surface area of Hinze Dam was set at 972ha, as per the specifications prior to the dam wall raising.

For the 'Diversified infrastructure' scenario, which incorporates the dam raising, the CH_4 flux estimate was increased by 1.24 times, being the square root of the 55% increase in Full Supply Level surface area after the dam wall was raised. This assumes that overall bubble-sourced emissions might increase in proportion to increases in the water body circumference, given the observation that bubble flux emissions occur predominantly at shallow depths at the periphery of the water body (Grinham et al. 2011).

Despite the availability of local empirical data, the estimates for dam-generated CH_4 should be considered as highly uncertain. There is scant information available to guide the prediction of CH_4 flux rates in response to changes in management strategy for an existing dam (such as the dam wall raising considered here). Furthermore, the available monitoring studies report gross emissions only, and there remain gaps in understanding what level of net anthropogenic emissions could be attributed to the existence of the water supply dams.

N2O from sewage treatment plants

The potential for notable levels of N_2O emissions from urban wastewater treatment systems has long been recognised (Czepiel et al. 1995). Until recently, however, there has only been scant information available on which to base the quantitative estimates required for greenhouse gas footprinting and LCA studies.

Recent years have seen a significant increase in research effort both to quantify N_2O emissions from STPs, and to understand the fundamental mechanisms leading to N_2O creation. This has exposed the inadequacies of conventional N_2O accounting methodologies. For example, a number of studies have estimated N_2O emission levels an order of magnitude higher than would be calculated using the default methodology proposed by the IPCC (see Daelman et al. 2013b, Kampschreur et al. 2009). In the Australian context, the mandated approach (DCCEE 2011) and recent literature (de Haas et al. 2009, Foley et al. 2010b) relate N_2O flux to the level of denitrification that is achieved, reflecting the historical perception that the denitrification pathway was the main source of N_2O from STPs. However, recent investigations have shown that process conditions affecting the nitrification pathway can be an important determinant of overall N_2O emission rates (Ahn et al. 2010, Law et al. 2012b, Ni et al. 2013, Sun et al. 2013, Ye et al. 2014). Denitrification based emission factors are therefore not likely to correlate consistently well with the key causative factors of N_2O generated in wastewater treatment processes.

There are now quite a number of published emission estimates for full scale STPs, covering a variety of process configurations and process conditions (see Desloover et al. 2012, Law et al. 2012b). While it is clear that the type of STP configuration could play an important role in determining the scale of N_2O emissions (Ahn et al. 2010, Foley et al. 2010b), there is not enough literature data available to meaningfully select emission estimates specific to the STPs included in this case study. Firstly, this is because the scale of emissions can also be strongly influenced by the variation in process operating conditions that will exist across similar installations (Ahn et al. 2010, Foley et al. 2010b, GWRC 2011, Law et al. 2012b). Secondly, recent studies have illustrated the extent to which N_2O fluxes can vary over diurnal to seasonal timescales (Aboobakar et al. 2013, Ahn et al. 2010, Daelman et al. 2013a, Sun et al. 2013, Ye et al. 2014) - casting doubts over the representativeness of most previous analysis based on short term sampling regimes. Others have identified another potential source of underreporting, claiming the physical sampling apparatus used in some previous studies was unlikely to have effectively captured N_2O from the major emission zones (Ye et al. 2014).

While the available data generated in recent years has clearly helped to highlight the potential scale of this emission source, it is somewhat unreliable for the purpose of proposing generic emission factors for large scale STPs (Aboobakar et al. 2013, Ahn et al. 2010, Daelman et al. 2013b). Given this, the estimate for this study was based on recent data collected across a large number of sites, rather than extracting information from any one particular example. Overall estimates were considered for 30 full scale, urban STPs, covering a range of process configurations and conditions (Aboobakar et al. 2013, Ahn et al. 2010, Daelman et al. 2013b, Foley et al. 2010b, GWRC 2011, Joss et al. 2009, Kampschreur et al. 2008, Ni et al. 2013, Sun et al. 2013, Ye et al. 2014). The average (15.5 g-N_2O/kg-TN in the influent) of those 30 flux estimates is double the median value (7.5 g-N_2O/kg-TN in the influent), illustrating the importance of recognising and understanding the variation in the published data. High uncertainty will be associated with any emission factors derived from the available data.

To allow for the possibility that some of the higher estimates in this set might reflect transient conditions that are not realistic over the longer term, the default emission factor was set at the median of the data surveyed (7.5 g-N_2O/kg-TN in the influent). However, given the concern that past sampling regimes could have underreported N_2O flux levels, it is also possible that this is excessively conservative on the low side. Clearly this value, or any other generalised emission factor, should be used for illustrative purposes only.

N2O from biosolids disposal

From the operations data collected for this study, 20-40% of the sewage nitrogen ends up in the biosolids waste stream. Disposal of these biosolids therefore represents the second largest flux of nitrogen leaving the sewage treatment system, and could potentially be another substantial pathway for N_2O generation.

The available literature indicates large variability in the possibility of N_2O generation from land-applied biosolids. Recommendations for the Canadian wastewater industry ranged from 8 to 36 g-N_2O per kg-N applied, depending on the soil type of the field to which the biosolids is applied (Brown et al. 2010). No Australian studies were found that measure actual N_2O generation from fields to which biosolids have been applied. However, local SEQ studies did identify that mineralisation rates of biosolids-N were much higher than is assumed by guidelines for calculating biosolids application rates (Barry and Bell 2006, Pu et al. 2008). This implies a higher risk of N_2O emissions than might otherwise be expected by the industry.

Biosolids use on agricultural fields can reduce the need for alternative fertiliser products, which themselves would have otherwise caused some degree of 'anthropogenic' N_2O emissions. As with biosolids use, the actual N_2O generation from synthetic fertiliser use could vary greatly depending on crop type, the choice of management practices, and climatic factors. In the Australian context, the potential for soil-generated N_2O emissions is considered to increase in warm or humid climates, and where crop irrigation is practiced (Thorburn et al. 2013). A recent review of Australian field data indicates that agricultural N_2O emissions can vary from less than 0.2% of applied N (substantially lower than the default IPCC value), to as high as 27% of applied N, depending on the crop type and location (Thorburn et al. 2013).

The case study analysis does not utilise literature based emission factors for field N_2O fluxes, because of (a) the lack of locally relevant guidelines that accounts for mineralisation rates of the (predominantly) organic nitrogen in biosolids; and (b) the difficulty in achieving an 'even' handling of the uncertainty across the two different fertiliser types.

Emission factors were instead taken from the IPCC guidelines for agricultural inventory (2006), which recommend using the same value (15.7 g-N_2O per kg-N applied to soils) for both mineral and organic nitrogenous fertilisers. The net N_2O flux attributed to the biosolids application therefore becomes dependent on the assumption used for bioavailability of the biosolids-N, which dictates the ratio of total biosolids-N (applied) to total synthetic fertiliser-N (avoided).

Clearly there is potential for large spatial and temporal variability in the net-N_2O flux attributed to the application of biosolids to farming soils. While conventional N_2O emission factors have been used in this study, these should be considered as highly uncertain and therefore most useful if taken as an indicative guide.

N2O from wastewater disposal

The default values for wastewater discharge to the ocean (3 g-N_2O per kg-N discharged), and via irrigation (12.6g-N_2O per kg-N discharged), are based on the median values identified in a previous literature review (Foley and Lant 2007).

Such estimates will be extremely uncertain because of a lack of targeted empirical data, particularly when used for estimating N_2O emission from tertiary treated wastewater streams with very low N concentrations. However, the effect of this on the greenhouse gas analysis of this study is likely to be minor. For the STPs in our scenarios, only 6**%** of the sewage nitrogen ends up in the wastewater, meaning that any change in N_2O emission factor for this particular process flow will have little effect on the overall N_2O balance.

This wastewater disposal pathway might have more significance in other regions that do not employ such advanced biological nutrient removal, particularly as urban populations grow and global oceans warm and deoxygenate (Short et al. 2013).

s3.5. Micropollutant assays

In most cases, detailed data was available to generate metals balances for the key streams of the various infrastructure components.

Much less directly-measured data was available for organic micropollutants, and the gaps were filled with locally relevant literature data where available. Where no such estimates were possible, assumptions were made to be conservative on the high side – i.e. tending towards higher organic concentrations in streams discharged to the environment.

STP product streams

For metal and organic micropollutants in secondary treated effluent, a generic contaminant profile was used for all the STPs. Metals concentrations were taken from STP discharge data, supplemented with extra analytes detected in available data from other SEQ STPs. No equivalent data was available for organic substances, so a hypothetical contaminants profile was developed by taking the maximum of measured data from two studies undertaken in the local region (Reungoat et al. 2010, Watkinson et al. 2009).

Statistical data on biosolids contaminant levels for the four STPs were provided by GCW, covering a range of metals and organic substances. A composite concentration profile was created from a weighted average of the mean contaminant concentration and the estimated 2007/08 dry solids production for each plant. This composite was then used as the default biosolids contaminant profile for each of the STPs. Additional data for molybdenum was taken from Foley et al (2010), as this was not included in the GC analytical data. Aluminium and Iron concentrations were calculated from the estimates of WTP sludge metals distributed to the sewer (see section s3.1), assuming these metals report entirely to biosolids.

Wastewater recycling - Direct non-potable reuse (DNPR) system

The metals balance for the UF-based DNPR system was based on measured data from the existing Pimpama-Coomera AWTP. This was used to estimate partitioning coefficients for each contaminant, calculated as the fraction of influent pollutant load that reaches the AWTP product water.

These partitioning coefficients were then applied to the generic secondary effluent contaminant profile in order to estimate the metal micropollutant flows in the DNPR system product water. We assumed that 100% of any removed metals would end up in the solid waste sludge stream. For organic micropollutants, we assumed no removal – hence, all organic micropollutants in the influent reported to the AWTP product water.

Wastewater recycling - Indirect potable reuse (IPR) system

The micropollutant (metals and organics) balance for the RO-based IPR system was based on data on micropollutant concentrations for the Bundamba AWTP influent and product water. From this we derived partitioning coefficients for each contaminant, calculated as the fraction of influent pollutant load that reaches the AWTP product water.

These partitioning coefficients were then applied to the generic secondary effluent contaminant profile in order to estimate the micropollutant flows in the product water for the hypothetical Coombabah AWTP system. For the micropollutant fractions removed from the product water, it was assumed that all metals report to the RO concentrate. Limited data was available to estimate the contribution to organics removal by each of the RO and product water oxidation steps. Unless the WaterSecure data indicated otherwise, a hypothetical 50% of the influent organics was directed to the RO concentrate, with the remainder of the removal attributed to the advanced oxidation process. This RO concentrate at the Bundamba facility is passed through a denitrification reactor before being discharged to a waterway. To be conservative, we did not allow for any oxidation of the concentrate organics during this treatment step. In our micropollutant balance, all metals and organic contaminants assigned to the RO concentrate were modelled as being discharged to the aquatic environment.

s3.6. Chemicals supply

Chemicals use by the different water system components are summarised in Table s10.

Manufacturing inventories were taken from the Life Cycle Strategies database (Grant 2012), otherwise from Ecoinvent data (Frischknecht et al. 2007). In either case, the majority of the manufacturing inventories are based on Ecoinvent v2 data, customised to use Australian electricity and other inputs. Assumed transport distances (for delivery to the Gold Coast city) are also provided in Table s10.

Locally specific inventories have been compiled for certain Australian manufactured chemicals (Alvarez-Gaitan et al. 2013), however the publicly available versions of that data are presented in the units of midpoint LCIA indicators, using different impact assessment models than those chosen for this study. It was therefore not possible to incorporate that data into the models used here.

The Alvarez-Gaitan et al. (2013) study highlights that product inventories based only on process databases (such as those used here) can substantially underestimate the full supply chain impacts of certain chemicals used by the water industry. For example, incorporating the more complete supply chain coverage that is possible when input-output based databases are used, increases the GHG footprint of Alum Sulphate (one of the higher volume chemicals used in this study) by 60%.

The importance of this methodological choice, and in using locally relevant production data, could be critical to LCA studies focussed on specific water cycle components (e.g. mains water treatment) where chemicals use is a major operational input. However, the effect on our study would be far less, since chemicals manufacturing makes a relatively small contribution to the overall LCIA results (e.g. <7% of the total GHG footprint). While improving the quality of the chemicals manufacturing inventories would be advantageous for urban water managers wishing to understand their life cycle environmental burdens, it would not change the generalised conclusions drawn in this study.

Table s10: Summary of chemicals usage data, provided as kg of bulk product per day. Solution strengths (as w-w %) are shown where appropriate. 'Product Water flow' for **the** *Sewage Treatment plants is equivalent to the average dry weather influent flow. 'TI mix' refers to the mix of infrastructure included in the 'Traditional Infrastructure' scenario.*

		Mains supply		Wastewater treatment		Wastewater recycling plants		Assumed transport distance (km)
		'TI' mix	Desal	'TI' mix	new plant (coupled with the DNR system)	IPR	DNR	
	Product Water flow (ML/d)	**137**	**125**	**108**	**17**	**44**	**17**	**--**
Chemicals usage (kg/d)	Methanol/Ethanol			80		220		1,000
	Alum Sulphate (46%)	8,750		1,570	3,560		370	500
	Ferric Sulphate (42%)		3,010					1,000
	Ferric Chloride (42%)					9,770		1,000
	Ferrous Chloride (28%)			1,320				1,000
	Lime (dry)	3,640		1,800	510			100
	Lime (hydrated)		3,750			1,540		100
	Polymer	200	370	140	50	30		1,700
	Sodium Hypochlorite (12%)	6,180	3,100	310	370	6,360	4,980	100
	Chlorine gas			560				100
	Carbon Dioxide gas	3,740	4,380			880		100
	Sodium Hydroxide (50%)	3,960				100		100
	Potassium permanganate	50						1,000
	Activated carbon	680						1,000
	Ammonium Sulphate (40%)					910		1,000
	Proprietary anti-scalant					90		1,700
	Citric Acid (50%)					60	10	1,000
	Hydrogen Peroxide (50%)					410		1,000
	Sodium Bisulphite (34%)		1,080			20		1,000
	Sulphuric Acid (98%)		5,330			970	10	1,000
	Total chemicals use	**27,200**	**21,020**	**5,780**	**4,490**	**21,360**	**5,370**	**--**

S4. ADDITIONAL DATA & RESULTS

s4.1. Summary of operational inventories

Table s11 provides more detailed description of the different technology types, and key operating inventories, for the two scenarios under evaluation.

Table s11: Summary of inventory flows for water supply and wastewater components

water system component		mains water supply			sewage collection & treatment			advanced wastewater treatment & recycling		rainwater tanks	
type		dam+WTP supplies		desal	default STP mix		extra STP	IPR	DNR	overall	
scenario		TI	DI	DI	TI	DI	DI	DI	DI	TI	DI
Treatment system		*Floculat'n; Filtrat'n; Disinfection*		*Filtration; RO*	*mixed nutrient removal technologies*			*Precip'n; MF; RO; AOP; ROC denitrific*	*Precip'n; UF*	*untreated*	
Flow balances											
Inflow	(ML/d)	129	179	298	108	278	17	54	17	--	--
Effluent	(ML/d)	--	--	--	87	170	0	0	10	--	--
Reject flow	(ML/d)	--	--	173	--	--	--	9	0	--	--
Product to treatment	(ML/d)	--	--	--	0	54	17	--	--	--	--
Product to users	(ML/d)	128	176	125	21	54	0	45	7	--	--
Rainwater yield	(ML/d)	--	--	--	--	--	--	--	--	3	59
Byproducts											
sludge generation	(t-ws/d)	1	3	--[a]	3	8	1	25[c]	0.4[c]	--	--
- destination		sewer		sea	Landfill			landfill	farms	--	--
- transport burden	(kt.km/d)	0	0	0	0.1	0.2	0.01	0.5	0.1	--	--
biosolids generation	(t-ws/d)	--	--	--	264	677	42	--	--	--	--
- moisture content	(%)	--	--	--	89%[b]	89%[b]	85%	--	--	--	--
- transport burden	(kt.km/d)	--	--	--	53	135	8	--	--	--	--
biogas generation	(ML/d)	--	--	--	3	8	0	--	--	--	--
Electricity use *- by user*											
influent supply	(MWh/d)	15	20	22	42	108	7	10	0	--	--
treatment	(MWh/d)	6	8	415	76	194	17	42	5	--	--
effluent discharge	(MWh/d)	--	--	--	20	40	0	3	4	--	--
product transfer	(MWh/d)	0	9	0	0	13	0	0	0	--	--
product to users	(MWh/d)	13	18	43	5	13	0	28	3	5	111
Electricity use - by source											
imported from grid	(MWh/d)	34	55	480	141	362	24	83	12	5	111
onsite generation	(MWh/d)	--	--	--	2	5	0	--	--	--	--
Chemicals use											
total usage	(t/d)	24	32	21	6	15	4	22	6	--	--
transport burden	(kt.km/d)	9	13	15	4	11	2	15	1	--	--
Direct gas emissions											
NH3	(kg-NH3/d)	--	--	--	385	1,067	78	0.0	0.6	--	--
N2O	(kg-N2O/d)	--	--	--	71	194	12	0.7	0.2	--	--
CH4	(kg-CH4/d)	506	610	--	643	1,727	18	--	--	--	--
CO2 non-biogenic	(kg-CO2/d)	--	--	--	7,718	20,678	1,172	572	119	--	--
Outflow concentrations											
TN / TP	(mg/L)	--[a]	--[a]	--[a]	3.9 / 3.1[b]	4.1 / 2.8[b]	1.6 / 0.3	0.3 / 0.02	1.0 / 0.2	--[a]	--[a]
total chlorine	(mg/L)	--[a]	--[a]	--[a]	0.3[b]	0.3[b]	--	0.0	1.2	--[a]	--[a]
Reject flow concentrations											
TN / TP	(mg/L)	--	--	--	--	--	--	13.5 / 0.9	--	--	--
total chlorine	(mg/L)	--	--	--	--	--	--	0.04	--	--	--
Biosolids concentrations											
TN / TP	(g/kg-ds)	--	--	--	71 / 39[b]	71 / 39[b]	70 / 37	--	--	--	--
TC	(g-C/kg-ds)	--	--	--	357[b]	357[b]	385	--	--	--	--
Displaced products											
rural water use	(ML/d)	--	--	--	7	7	0	0	0	--	--
Urea	(t/d)	--	--	--	1.2	3.0	0.3	--	--	--	--
DAP	(t/d)	--	--	--	2.9	7.4	0.6	--	--	--	--
fertiliser transport	(kt.km/d)	--	--	--	0.21	0.55	0.04	--	--	--	--

[a] *not assessed*
[b] *flow-weighted average for the STPs included in this mix*
[c] *includes chemical sludge + biologically generated solids*

s4.2. Additional LCIA results

Table s12 provides the absolute results for the two scenarios. Table s13 breaks those overall results down by different components of the urban water system.

Table s12: Impact results for the Traditional & Diversified scenarios, showing annualised results in absolute units.

	FES GL-eq	MEU t-eq N	FEU t-eq P	MTX t-eq DCB	FTX t-eq DCB	TTX t-eq DCB	TA t-eq SO2	GW kt-eq CO2	OD kg-eq CFC11	FFD kt-eq oil	MD Mt-eq Sb	HTX kt-eq DCB	POF t-eq NMV	PF t-eq PM10
'Traditional infrastructure'	45.1	931	10	219	12	47	834	103	578	22	326	8	403	197
'Diversified infrastructure'	46.4	1,858	30	544	45	129	3,493	487	1,828	114	1,042	27	2,104	922

Table s13: Breakdown (as % contributions) of the overall Impact Category results for each scenario (shown as Traditional / Diversified), by the main components of the urban water system. The rows in grey provide subtotals, and match the breakdown classification used in Figure 2. Contributions are only shown if they are ≥1% in a positive or negative direction, hence some of the subtotals include results that do not appear in this table.

		FES (%)	MEU (%)	FEU (%)	MTX (%)	FTX (%)	TTX (%)	TA (%)	GW (%)	OD (%)	FFD (%)	MD (%)	HTX (%)	POF (%)	PF (%)
Operations	Dams	104 / 101							4 / 1						
	WTP			3 / 1	4 / 2	35 / 14		13 / 4	13 / 4	2 / 1	18 / 5	21 / 9	13 / 6	15 / 4	15 / 4
	Desal		0 / 1	0 / 2	0 / 4	0 / 14		0 / 25	0 / 31	0 / 9	0 / 33	0 / 20	0 / 13	0 / 33	0 / 30
	Mains water distrib'n					0 / 1		3 / 4	4 / 5	1 / 1	5 / 5	2 / 3	1 / 1	5 / 5	4 / 5
	Mains water supply	104 / 102	0 / 1	3 / 4	5 / 7	36 / 29	0 / 1	16 / 33	22 / 40	3 / 11	22 / 42	23 / 31	14 / 19	20 / 42	19 / 39
	Rainwater supply				0 / 1	0 / 1		1 / 6	2 / 8	0 / 2	2 / 9	1 / 4	0 / 2	2 / 9	2 / 8
	Sewage collection				1 / 1	1 / 1		10 / 6	19 / 11	3 / 2	16 / 8	5 / 4	3 / 2	17 / 9	13 / 7
	Sewage treatment		100 / 93	95 / 93	88 / 71	43 / 33	92 / 90	65 / 42	43 / 24	92 / 79	39 / 20	15 / 12	55 / 45	41 / 20	54 / 31
	Sewage collection & treatment		100 / 94	95 / 93	89 / 72	45 / 34	92 / 91	74 / 49	62 / 35	94 / 82	55 / 28	20 / 17	58 / 48	57 / 29	67 / 39
	WW recycling	-6 / -5	0 / 5	0 / 2	0 / 14	5 / 27	7 / 8	0 / 7	1 / 9	0 / 3	1 / 9	0 / 8	2 / 10	1 / 9	1 / 8
Construction		1 / 3	0 / 0	2 / 1	6 / 6	15 / 9	1 / 1	8 / 6	13 / 8	2 / 2	19 / 12	56 / 40	26 / 21	19 / 11	11 / 7
other															
Total		100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%

S5. IMPACT MODEL SELECTION

This section provides more detail on the impact assessment models chosen for five specific impact category types. Two of those relate to the issues most relevant to current environmental management strategy for the urban water industry in Australia – freshwater use (*Freshwater Extraction Stress*) and nutrient discharge (*Eutrophication*). The model descriptions provided for the other three issues (*Ecotoxicity Ozone, Minerals Depletion*) underpin the detailed critique provided in the main paper.

s5.1. Freshwater Extraction Stress

Historically, assessment of water use remained one of the least well developed methodologies in LCA research. While a number of different approaches have attempted to rectify this gap in recent years, no consensus has been reached on a preferred methodology for use in LCA (Kounina et al. 2013).

The Water Stress Index (WSI) approach proposed by Pfister *et al.* (2009) is adopted here. That metric acknowledges the importance of accounting for the sensitivity of different watercourses to a unit amount of freshwater extraction, providing WSI values that reflect regional differences in the ratio of extraction to availability, and rainfall variability. Of those recent methodologies proposed for use in midpoint LCA, this is one of the more widely applied in the literature.

A WSI characterisation factor of 0.065 was used for all stream extractions related to the provision of water to urban users in the Gold Coast region. This was taken as the unweighted average of those grid cells in the Pfister *et al.* (2009) model that coincide with the catchment boundary of the Gold Coast rivers.

A more generalised approach was used for applying WSI characterisation factors to water use at other stages of the system life cycle. While spatially resolved WSI values exist for all regions of the world, they could not be utilised fully because the background life cycle inventory databases used in this study contained insufficient spatial information to determine the location of various freshwater extractions. To overcome this, it was assumed that the majority of water extraction associated with the supply chain of these Gold Coast scenarios will be occurring in Australian urban centres. All water extraction in the background inventories was assigned with a single WSI characterisation factor of 0.116. This was determined by taking the spatially-weighted average of the published WSI grid cells (Pfister et al. 2009) across all Australian regions defined as “Major Cities” or “Inner Regional” in the 2011 Australian Census (ABS 2013).

Both these WSI factors were normalised against the Gold Coast value, so as to provide the following characterisation factors:

- [1.0 ML-Gold Coast equivalent / ML of water extracted] applied to all Gold Coast dam extractions occurring in our urban water scenarios, along with the stream-sourced irrigation that is avoided via wastewater reuse.
- [1.8 ML-Gold Coast equivalent / ML of water extracted] applied to all other supply chain water use

The implication of using these characterisation factors is that water extraction in the Gold Coast catchment is relatively benign, when compared to water extraction taking place in catchments feeding the other major urban centres of Australia. Further investigation would be required to determine whether or not that conclusion is supported by the wealth of knowledge available on freshwater ecosystem condition in Australian river systems (Kennard et al. 2010).

Exclusions

The construction of water supply dams not only enables the intensive extraction of water for anthropogenic purposes, it also increases the rate of evaporative losses from the watercourse. This represents another substantial reduction in the water available to provide ecosystem services, and would therefore be a valid intervention to include in LCA analysis of urban water systems. However,

evaporative losses were excluded from our analysis, because they are also absent from much of the background inventory data that was available for use in this study.

Rainwater interception by Gold Coast household tanks was also excluded from the extraction inventories. This is justified on the grounds that, for the coastal Gold Coast City area, the vast majority of such interception will be diverting runoff that would otherwise flow directly to an estuarine environment. The rainwater tanks in this particular case study are therefore not expected to affect those major freshwater systems that are addressed by the WSI metric, and constitute the major focus of local environmental protection priorities (QG 2009).

s5.2. Eutrophication

Early metrics for *Eutrophication* in the LCA domain tended to sum all fluxes of nitrogen (N) and phosphorus (P), comparing the two on the basis of the Redfield ratio (Redfield et al. 1963) – a standard approach to ascertaining the relative N and P uptake by waterborne algae. More recent approaches have acknowledged the significant spatial variability in N and P status that exists across different waterways, and therefore the possibility that only one of the nutrients might be rate limiting to algal growth in many cases (Azevedo et al. 2013, Bare et al. 2006, Basset-Mens et al. 2006, Gallego et al. 2010, Goedkoop et al. 2009, Helmes et al. 2012, Hirosaki et al. 2002, Struijs et al. 2011). There is no equivalent *Eutrophication* model available to inform the application of LCA across Australia.

Table s14: Derivation of Marine Eutrophication Potential (MEU) characterisation factors for each type of emission; showing: (a) the fate factor for marine waters, and the specific oxidation potential of the contaminant; and (b) the overall characterisation factor, which represents the oxidation potential for each chemical, relative to the highlighted reference emission (an emission of nitrogen direct to the sea)

(a) Emission		fate factor (w/w fraction)[a]						specific oxidation potential [b] (kg-O2 / kg)		
		wastewater discharge to GC waterways			flux from SEQ agri-soils to inland streams	other life cycle emissions				
		dam	streams	estuary & sea		freshwater systems	marine systems	1°	2°	total
NH_4^+	(kg-NH_4^+)	0.8	0.8	1.0	0	0.7	1.0	3.6	15.0	18.6
other N species	(kg-N)	0.8	0.8	1.0	0	0.7	1.0	--	19.4	19.4
P species	(kg-P)	0.4	0.4	1.0	0	0	0	--	140	140
COD	(kg-COD)	0.0	0.0	1.0	0	0	0	1.0	--	1.0

[a] *mass fraction (kg/kg) of emissions reaching a marine waterbody in which it will stimulate a response*
[b] *mass of O2 depleted (kg-O_2) per mass of agent (kg) reaching the marine waterbody (Karrman and Jonsson 2001)*

(b) emission		oxidation potential (kg-O_2/kg emitted)					
		wastewater discharge to GC waterways			flux from SEQ agri-soils to inland streams	other life cycle emissions	
		dam	streams	estuary & sea		freshwater systems	marine systems
NH_4^+	(kg-NH_4^+)	14.9	14.9	18.6	--	13.0	18.6
other N species	(kg-N)	15.5	15.5	19.4	--	13.6	19.4
P species	(kg-P)	56	56	140	--	--	--
COD	(kg-COD)	--	--	1.0	--	--	--

emission		characterisation factor (kg-N-eq/kg emitted)					
		wastewater discharge to GC waterways			flux from SEQ agri-soils to inland streams	other life cycle emissions	
		dam	streams	estuary & sea		freshwater systems	marine systems
NH_4^+	(kg-NH_4^+)	0.77	0.77	0.96	--	0.67	0.96
other N species	(kg-N)	0.80	0.80	1.00	--	0.70	1
P species	(kg-P)	2.89	2.89	7.23	--	--	--
COD	(kg-COD)	--	--	0.05	--	--	--

Table s15: Derivation of Freshwater Eutrophication Potential (FEU) characterisation factors for each type of emission; showing: (a) the fate factor for freshwater streams, and the specific oxidation potential of the contaminant; and (b) the overall characterisation factor, which represents the oxidation potential for each chemical, relative to the highlighted reference emission (an emission of phosphorus direct to a freshwater stream)

(a) emission		fate factor (w/w fraction)[a]						specific oxidation potential [b] (kg-O2 / kg)		
		wastewater discharge to GC waterways			flux from SEQ agri-soils to inland streams	other life cycle emissions				
		dam	streams	estuary & sea		freshwater systems	marine systems	1°	2°	total
NH_3	(kg-NH_3)	0.2	0.2	0	0	0	0	3.6	15.0	18.6
other N species	(kg-N)	0.2	0.2	0	0	0	0	--	19.4	19.4
P species	(kg-P)	0.6	0.6	0	1.0	1.0	0	--	140	140
COD	(kg-COD)	1.0	1.0	0	1.0	1.0	0	1.0	--	1.0

[a] *mass fraction (kg/kg) of emissions reaching a freshwater system in which it will stimulate a response*
[b] *mass of O2 depleted (kg-O_2) per mass of agent (kg) reaching the freshwater system (Karrman and Jonsson 2001)*

(b) emission		oxidation potential (kg-O_2/kg emitted)					
		wastewater discharge to GC waterways			flux from SEQ agri-soils to inland streams	other life cycle emissions	
		dam	streams	estuary & sea		freshwater systems	marine systems
NH_3	(kg-NH_3)	3.7	3.7	--	--	--	--
other N species	(kg-N)	3.9	3.9	--	--	--	--
P species	(kg-P)	84	84	--	140	140	--
COD	(kg-COD)	1.0	1.0	--	1.0	1.0	--

emission		characterisation factor (kg-P-eq/kg emitted)					
		wastewater discharge to GC waterways			flux from SEQ agri-soils to inland streams	other life cycle emissions	
		dam	streams	estuary & sea		freshwater systems	marine systems
NH_3	(kg-NH_3)	0.03	0.03	--	--	--	--
other N species	(kg-N)	0.03	0.03	--	--	--	--
P species	(kg-P)	0.60	0.60	--	1.00	1	--
COD	(kg-COD)	0.01	0.01	--	0.01	0.01	--

Both N and P are considered to pose risks to the ecological health of freshwater streams, estuaries and coastal waterways of the SEQ region (Abal et al. 2005). For wastewater discharges and irrigation losses occurring in the Gold Coast catchment, we therefore chose to account for all emissions of N and P (using the Redfield Ratio for N vs. P equivalency) in the *Eutrophication* model. For all other nutrient emissions across the system life cycle, the affected water bodies were assumed to be either P-limited freshwater streams or N-limited marine waters. This is consistent with the approach promoted by recent LCA impact assessment methods (e.g. Goedkoop et al. 2009).

For all receiving waters, the *Eutrophication* models used here also account for the primary oxidation effects associated with both inputs of organic matter, and the nitrification of ammonia (following Karrman and Jonsson 2001). Doing so implies that the focus of the 'Eutrophication' metrics is on the potential for waterbody oxygen levels to be depleted – in the context of nutrient inputs, this would be expected as nutrient-stimulated algae subsequently degrades. Others have cautioned against such a hybridisation, given oxygen depletion is only one of the possible pathways from nutrient discharge to ecological degradation (ECJRC 2010). However, we feel this to be a worthwhile addition for this particular case study, given the strong focus around Australia on minimising COD and NH_4^+ inputs to waterways.

Table s14 and Table s15 summarise the derivation of Marine (MEU) and *Freshwater (FEU) Eutrophication* characterisation factors used in this study. These are based on the following assumptions for the different nutrient disposal pathways across the system life cycle:

- The Gold Coast STPs discharge their treated wastewater directly into the marine environment.

- For nutrients in the Indirect Potable Reuse system discharge to the water supply dam, 40% of P inputs and 80% of N inputs would transfer downstream from the dam and eventually to the sea. These delivery ratios are informed by nutrient budgets developed for non-flood years in a major water supply dam system of a neighbouring region (Burford et al. 2012).
- Nutrient losses from wastewater irrigation were assumed to reach tidal sections of the streams, and then be transferred to marine ecosystems.
- For all other life cycle nitrogen inputs to freshwater streams, it is assumed that 70% will eventually reach a marine environment. This is taken from the default delivery ratios embedded in the ReCiPe models for eutrophication assessment (Goedkoop et al. 2009).

Nutrient losses from biosolids and synthetic fertilisers into inland streams were assumed to have no effect on the marine environment. The disposal zone considered for biosolids reuse is at the very head of the Murray-Darling catchment of Australia, which discharges into the ocean approximately 2000km to the south. It is assumed here that the transfer of nutrients to the river mouth would be negligible, given large sections of that downstream river system are highly episodic, and substantial deposition of sediments onto floodplains can occur. For the emission of nitrogen species into airsheds, fate factors were taken from the European-based values used in the ReCiPe model (Goedkoop et al. 2009), as no equivalent data has been collated for use in Australian LCA applications.

s5.3. Ozone Depletion

It remains common practice (e.g. Goedkoop et al. 2009, Hauschild et al. 2013) in LCA application to use characterisation factors for *Ozone Depletion (OD)* assessment based on the 2002 or 2006 scientific reviews on the state of the ozone layer for the World Meteorological Organization (WMO 2003, 2007). A number of those Ozone Depletion Potential (ODP) factors have since been revised again in the 2010 assessment report (WMO 2011).

Characterisation factors (Table s16) for halocarbon substances were taken from the semi-empirical, steady-state ODP values updated in that most recent WMO review of the science (Daniel et al. 2011).

Table s16: Ozone Depletion Potential (ODP) characterisation factors used in this study

WMO classification	Substance	ODP (kg-CFC11-eq/kg)	Source
Annex A-I	CFC-11	1	
	CFC-12	0.82	*[a]*
	CFC-113	0.85	*[a]*
	CFC-114	0.58	*[a]*
	CFC-115	0.57	*[a]*
Annex A-II	Halon-1301	15.90	*[a]*
	Halon-1211	7.90	*[a]*
	Halon-2402	13.00	*[a]*
Annex B-II	CCl_4	0.82	*[a]*
Annex B-III	CH_3CCl_3	0.16	*[a]*
Annex C-I	HCFC-22	0.04	*[a]*
	HCFC-123	0.01	*[a]*
	HCFC-124	0.02	*[c]*
	HCFC-141b	0.12	*[c]*
	HCFC-142b	0.06	*[a]*
	HCFC-225ca	0.02	*[c]*
	HCFC-225cb	0.03	*[c]*
Annex E	CH_3Br	0.66	*[a]*
Others	Halon-1202	1.70	*[b]*
	CH_3Cl	0.02	*[a]*
	N_2O	0.018	*[d]*

For each halocarbon, the WMO assessment that contains the most recent ODP update is listed: [a] WMO (2011); [b] WMO (2007); [c] WMO (2003).

[d] ODP factor for N_2O from Lane and Lant (2012)

N_2O emissions are also included in the assessment provided here, given anthropogenic N_2O sources are expected to be the biggest agent of ozone layer depletion over the next century (Portmann et al. 2012, Ravishankara et al. 2009). A steady-state ODP factor of 0.018 kg-CFC11-eq / kg-N_2O is used, following the rationale that this can provide useful insight for analysis of systems where N_2O has a prominent role (Lane and Lant 2012). The three calculated ODP values in the literature agree closely (0.017 – 0.019 kg-CFC11-eq / kg-N_2O), all being modelled using a consistent set of atmospheric conditions (Ravishankara et al. 2009; Daniel et al. 2010; Fleming et al. 2011).

It should be noted, however, that some caution is required when interpreting results that integrate this non-halocarbon species into the conventional *OD* assessment framework. The modelling approaches used to calculate the ozone depletion potential of N_2O (as above) are conceptually compatible with the more conventional ODP values derived for halocarbon substances (see Daniel et al. 2011, Solomon et al. 1992). However, the strong level of scientific support for the ODP concept is in part due to the fact that any inherent uncertainties in the available atmospheric models will affect most halocarbon species in a similar manner. Much of that uncertainty would cancel out when halocarbon ODP values are calculated relative to a reference substance. Compared to halocarbons, there are certain fundamental differences in the spatial and temporal characteristics of the interaction between N_2O and stratospheric ozone. As a result, the risk of unrecognised modelling bias is increased, when both N_2O and halocarbons are considered and compared using the OD metric (see Fleming et al. 2011).

An important practical consideration is that the relative importance of N_2O vs halocarbon emissions might change, depending on assumptions made over the future evolution of greenhouse gas and halocarbon emission rates. The level of ozone damage caused by N_2O is very sensitive to background stratospheric conditions, particularly temperature (which is strongly influenced by atmospheric CO_2 levels) and chlorine abundance (determined by halocarbon emission rates) (Fleming et al. 2011, Plummer et al. 2010, Portmann et al. 2012, Revell et al. 2012). Likewise, the ozone destructiveness of chlorinated halocarbons will be sensitive to background CH_4 and N_2O emission rates, and also temperature. These non-linear interactions are expected to change substantially, and to very different effect for the various ozone depleting agents, over the course of the next century.

Clearly there remain some challenges in resolving the best way forward on the issue of N_2O and ozone depletion. However, sticking with the more conventional approach would risk the perverse outcome whereby the use of LCA would reinforce, rather than challenge, the common perspective that the ozone depletion problem has largely been 'fixed'. Conversely, including the available ODP factors for N_2O would allow decision makers to test the sensitivity of their LCA results to this issue, with the confidence that their approach reflects the best available scientific thinking. Until a clearer direction is provided by the international science and policy communities, the latter approach is recommended if LCA studies are to include metrics of ozone layer depletion.

s5.4. Minerals Depletion

The *Minerals Depletion* assessment in this study has a particular focus on exploring the significance of recovering phosphorus from the wastewater system. Phosphorus is a critical, non-substitutable, ingredient in food production, and global demand is forecast to grow substantially over the next century (Van Vuuren et al. 2010). This is expected to drive up the price of mineral phosphorus fertilisers, increasing the financial incentives for phosphorus recovery from urban wastewater systems (Sartorius et al. 2012, Van Vuuren et al. 2010, von Horn and Sartorius 2009). While sewage phosphorus constitutes only a small portion of overall global and national-level P flows (Chowdhury et al. 2014), its recovery does offer the opportunity to buffer local regions against future phosphorus price increases (Cordell et al. 2013).

Of the many LCA models that purport to assess mineral resource sustainability, only three were found that include characterisation factors for mineral phosphorus resources, and are therefore capable of incorporating biosolids phosphorus recovery into the overall analytical framework. One of those – the EPS2000 metric (Steen 1999) – relies on an assessment of implications in the distance future when concentrated ore bodies no longer exist. This approach appears to have gained little traction in the LCA research community. The other two – CML-IA (van Oers et al. 2002) and EDIP (Hauschild and Potting 2005) – have been widely adopted in applied LCA literature. The latter two approaches are structurally very similar, both employing simplified metrics based on ratios of usage : stocks. Of the three options, the CML-IA metric incorporates the most recently updated data on minerals resource stocks.

The *Minerals Depletion* model applied in this study uses characterisation factors for mineral resources taken from version 4.2 of the CML-IA method for *Abiotic Depletion Potential* (CML 2013).

That *Abiotic Depletion Potential* model integrates both mineral and fossil fuel resources (see van Oers et al. 2002), however the latter were not used in this study. The issue of fossil fuel resource depletion is captured here under a separate impact assessment category.

The CML-IA methodology for *Abiotic Depletion Potential* includes three possible metric alternatives, with each essentially addressing a different timeframe of concern (van Oers et al. 2002). The 'ultimate reserve' approach takes the longest term view, benchmarking the depletion against estimates of the total mineral stock in the earth's crust. At the other end of the spectrum is the 'economic reserve' approach, which benchmarks the depletion against the stock levels currently considered to be economically viable to extract.

The 'economic reserve' metric is used in this analysis. Of the three CML-IA possibilities, this is the one most closely aligned with the nature of the problem pertaining to future phosphate rock availability. That judgement assumes that the most pressing risk related to phosphate resource availability is the potential for future price rises to compromise food production in poorer regions of the world. In that context, it is the shorter term supply-demand benchmarks of the 'economic reserve' metric that are of the greatest interest.

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APPENDIX C1 CASE STUDY #2 – BIOSOLIDS AND AGRICULTURE – A GROWING 'LCA RISK' FOR URBAN WATER UTILITIES

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BIOSOLIDS AND AGRICULTURE - A GROWING 'LCA RISK' FOR URBAN WATER UTILITIES

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ABSTRACT

60% of Australian sewage treatment plant (STP) biosolids are applied to farm soils as a fertiliser supplement, helping to alleviate a substantial waste management challenge for urban water utilities. Following a decade long research program which concluded that the risks associated with biosolids application to Australian soils are minor, the water industry is actively promoting that this practice be continued and expanded. At the same time, the Australian agricultural industry is embracing the use of the LCA methodology to support their products, and it is expected that the influence of LCA on farm-scale decision making will grow. The application of LCA to agricultural systems is inherently complex, and it is in the water industry's interest to ensure that LCA of biosolids can adequately characterise the pros and cons of STP biosolids as a viable farming input.

To explore how agriculture-focussed LCA might affect the water industry, a simplified case study is reviewed in two ways – firstly, focussing on the modelling of impacts associated with field application of biosolids; and secondly, by quantifying the net implications of a change in biosolids disposal practice. Sensitivity analysis is used to explore the significance of parameters that can vary substantially depending on case study circumstances. Literature review was used to underpin the modelling of different input parameters, also to select and customise the best available life-cycle impact assessment (LCIA) models for application to biosolids analysis.

Shifting the point of biosolids disposal, from landfill to farm soils, substantially worsens the 'environmental footprint' of an Australian grain production system, if measured using contemporary LCIA models. However, the LCA characterisation of key risks (heavy metals loadings to soil; nutrient flux to waterways) and benefits (phosphorus recovery) do not accord with local scientific assessment of the hazards involved, and will tend to bias against the choice of biosolids as a fertiliser input. In other cases, particularly in terms of greenhouse gas (GHG) accounting, the barriers to impact quantification are more to do with gaps in the available knowledge base.

If the growth of agriculturally focussed LCA extends to the consideration of alternative fertiliser inputs such as STP biosolids, this could have substantial ramifications for Australia's urban water industry. Analysis driven by the needs of farmers might require a very different set of methodological approaches (system boundaries; inventory assumptions; impact categories) than have been employed in past water-industry sponsored analysis of biosolids disposal options. It will also become increasingly likely that such analysis will be conducted by practitioners who are unaware of the breadth of contemporary scientific wisdom on the low risks associated with biosolids application to Australian soils. If such developments were to reduce farmer enthusiasm for utilising biosolids as a crop supplement, this could pose a substantial risk over the medium to longer term for those Australian water utilities already heavily reliant on the agricultural disposal option.

INTRODUCTION

The water industry perspective

Biological treatment of municipal wastewater generates large volumes of waste solids, creating a perennial challenge for the urban water industry. Given the global trend to urban consolidation, this challenge is only likely to grow (LeBlanc et al. 2008). In Australia, ~60% of the 330kt/yr of waste biosolids produced from sewage treatment plants (STPs) are applied directly onto agricultural fields (ANZBP 2013). This helps to alleviate the waste disposal problem for water utilities, while providing a source of nutrients and organic matter for the farmers involved. Similar

practices are common, albeit to varying extents, in many other countries across the world (LeBlanc et al. 2008).

In Australia, a coordinated research program into the effects of biosolids use for agriculture has been underway for more than a decade (McLaughlin et al. 2007, Priest et al. 2012). Those investigations have concluded that appropriately managed biosolids use can deliver benefits at the field (e.g. improved soil condition & crop yields), at low risk to human health and the local environment (Pritchard et al. 2010). The water industry has extended those arguments in national policy debate, highlighting also the opportunity for GHG credits via the sequestration of carbon in the farm soils, and the broader benefits of capturing the nutrient resources contained in biosolids (AWA 2012). Collectively, these endeavours have underpinned enthusiasm to expand the opportunities for agricultural use of biosolids, as urban water utilities look to avoid the risks inherent in relying on landfills to continue taking such large volumes of STP waste product.

The agricultural industry perspective

In Australia's agricultural industry, certain independent developments are underway that might one day have some bearing on biosolids management. With so many of Australia's agricultural sectors being dependent on exports to foreign countries, the industry is embracing the use of Life Cycle Assessment (LCA) as an analytical tool. Internationally, there is a strong trend towards the use of LCA to support decisions made on the choice of food imports. Driving this is a mix of consumption-based national GHG initiatives (Barrett et al. 2013), more general policy imperatives (Wolf et al. 2012), high profile (albeit sometimes transient) consumer concerns (e.g. food miles), and the more substantial shift towards incorporating metrics of 'environmental footprint' on food product labelling.

Responding to these pressures, the Australian agricultural sector has recently finished the first stage in a research program to standardise, and build capacity in, the application of LCA to Australian agriculture (Eady et al. 2012, Eady et al. 2014). Outputs from that project are designed for ease of use, and will be widely accessible, enabling the use of LCA for agricultural analysis by a much greater range of parties than is currently possible. As this endeavour gathers momentum, it is expected that LCA will become widely used to inform the decisions of farmers who have an eye on the export market.

LCA and the use of biosolids for farming

At first glance, LCA would seem well suited to assessing the bigger picture implications of agricultural use of waste biosolids. Backed by ISO standards (ISO 2006), and underpinned by widely accessible and ever-improving inventory databases, the LCA methodology is designed to encourage rigour and transparency in the analysis of tradeoffs that occur at dispersed parts of supply chain life cycles (Bauman and Tillman 2004). Furthermore, there is a long history of LCA being used for analysis of a broad spectrum of environmental issues (e.g. aquatic eutrophication; toxic impacts; climate change; resource depletion) that overlap with those featuring most prominently in the scientific and policy arguments in favour of biosolids use on farms.

However, the results from such broad-spectrum LCA would seem at odds with the water industry's enthusiasm for agricultural use of biosolids. Previous Australian studies suggest that this can contribute substantially to the overall life-cycle burden of the urban water sector; albeit noting reservations about the utility of the available life-cycle impact assessment (LCIA) models for reaching such a conclusion (Lane et al. under review, Lundie et al. 2004).

The majority of LCA applied to biosolids has in fact been from the perspective of the water industry, used specifically to compare different biosolids disposal options (see Yoshida et al. 2013). In that body of work, there has been very little critique applied to how well the issues relevant to agricultural use of biosolids can actually be characterised. Instead, the majority of analytical focus is directed towards engineered treatment and disposal technologies for managing the solids stream (Yoshida et al. 2013).

Risks for the water industry…?

Taken at face value, this suggests the possibility of a growing risk for urban water utilities. In Australia at least, farmers have until now been generally quite willing to utilise STP biosolids where they are available – they are typically provided at no cost, and save them expenditure on conventional fertilisers. However, if LCA gains traction as a decision support tool for use in the agricultural sector, then closer attention might be paid to whether or not biosolids use compares favourably on environmental grounds, when compared with the use of alternative fertiliser sources.

To explore the risk this might pose to Australia's water industry, this study reviews a number of commonly used LCA modelling assumptions, and LCIA models, in key areas of relevance to biosolids analysis. A case study, simple in design but incorporating novel features for LCA analysis of biosolids disposal, is used to consider the (mis)alignment of these approaches with the best available science from Australian biosolids research. The results highlight the challenges in using the LCA methodology for analysis of agricultural biosolids use, considered from the perspectives of both the urban water and agricultural sectors.

METHODOLOGY

Overview of the case study analysis

The case study is set in the context of the biosolids management of a particular Australian water utility (Lane et al. under review). In that previous study, a city-scale urban water system (including water supply and wastewater treatment) is analysed across a diverse range of LCIA categories, using a functional unit of '*the provision of water supply and wastewater management services, for a one year period, to an urban population in the Gold Coast region of Australia*'. That modelling assumed that 100% of the biosolids generated by wastewater treatment is directed to use on farmlands.

For the analysis provided in this appendix, the agricultural application system is modelled on field data from a specific trial study, providing a consistent set of assumptions regarding the interaction between nutrient balances and crop response. To this end, data was taken from the 2002/03 trials at a property near the Cecil Plains township in South East Queensland (Barry and Bell 2006). That study compared the soil and yield response for two fields growing sorghum crops – one fertilised with biosolids, and the other using conventional fertiliser products.

The analysis for this case study is presented in five steps:

1. Firstly, an initial review is undertaken into the GHG implications of biosolids application to farm soils.
2. Other key inventory uncertainties are also considered, focussing here on assumptions associated with the soil nutrient balance.
3. Two particular analytical challenges are considered from the perspective of the impact assessment models employed in LCA.
4. The results are then considered in the context of a change in management approach from biosolids disposed in landfill, to biosolids used on farms. Such a change is not just relevant for decision making in the urban water industry, but also for the agricultural sector. Biosolids generation rates will typically be dictated by STP effluent quality requirements, and the influence this has on the choice of STP treatment technologies. Because of this, the decision by a farmer to begin (or cease) using biosolids as a soil supplement will influence the disposal pathway chosen for that biosolids, rather than the amount of biosolids produced.

   Landfill was chosen as the default disposal method, since this is current practice for many small to medium sized urban water utilities around Australia. While some of the largest Australian utilities instead stockpile their waste biosolids, CH_4 fluxes from such stockpiles (e.g. Majumder et al. 2014) are likely to be less than those estimated using the first-order decay model applied to the landfill disposal scenario, hence the GHG implications would fall within the range of landfill-based estimates considered in this study.

5. In order to provide some perspective on the tradeoffs involved in the change of disposal approach, the results are benchmarked in two ways. Firstly, they are compared against the equivalent LCA burden associated with operating a wastewater and water supply system of an equivalent scale. This provides the water industry perspective – what does the disposal change mean for their environmental footprint. To provide the equivalent insight from the farmer's perspective, the results are also critiqued using an overall grain cropping system benchmark. The modelling of all benchmarks is described further in the supplementary information (Appendix C2).

Scenario definition

System boundary for the analysis

So as to focus primarily on the agricultural-use aspect of the biosolids system, two important simplifications are made in defining the system boundary of this study.

Firstly, any (potential) need for additional STP processing to stabilise or dewater the biosolids is excluded, essentially assuming that the STP waste product in the baseline (landfill) scenario already meets the local quality standards for application to agricultural fields. Depending on the technology employed, such treatment could be a substantial component of the LCA 'burden' for the overall biosolids system (Peters and Rowley 2009); and should therefore be included in analysis underpinning actual decision making processes in specific locations.

Secondly, the analysis is also limited to changes in system operations, on the premise that minimal additional on-farm physical infrastructure would be required to facilitate the agricultural use. The possibility of changes in the physical infrastructure or land footprint required for urban landfill is also not considered. Past studies have found the infrastructure construction life-cycle stage to make only minor contribution to the LCA results for similar systems (e.g. Peters and Rowley 2009).

The system boundary does incorporate the implications of offsets generated by the disposal of the biosolids. For the landfill option, this includes the methane produced from degradation of the biosolids carbon, then used to generate electricity. For the agricultural use option, (a) the reduced need for manufacture and supply of synthetic fertilisers is credited to the final results; and (b) a credit was given for crop yield increases attributed to the biosolids application.

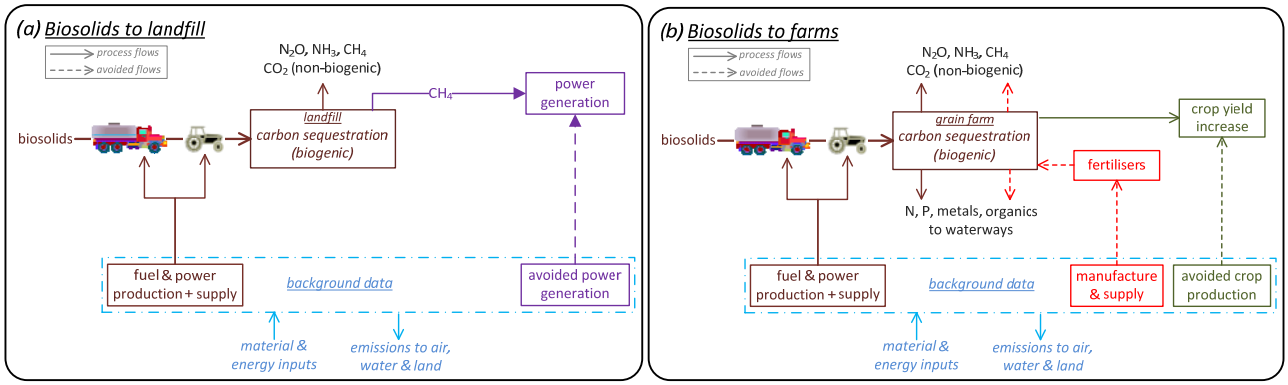

Figure 1: System boundary for the scenarios involving (a) disposal in an urban landfill site; and (b) direct application of biosolids to agricultural land. Solid lines represent flows that are attributed to the biosolids disposal. Dotted lines represent flows that are avoided by the use of biosolids, and allocated as a credit to the scenario results.

Biosolids generation

The biosolids generation rate and composition (Table 1; Table 2) are modelled on the overall 2008 production from four STPs in the Gold Coast region of Australia, with the composition (moisture, N, P, C, trace metals and organics) is set to equal the average (weighted by total mass of dry solids) of the available data for those four STPs (Lane et al. under review). All solids generation in that particular case study satisfied local regulatory requirements for application to farm soils.

Table 1: Biosolids generation characteristics

STP throughput (ADWF)	(ML/d)	108
biosolids generation	(t-ds/d)	30
biosolids composition		
moisture	(w/w%)	89
total nitrogen	(kg-N/t-ds)	71
phosphorus (biologically bound)	(kg-P/t-ds)	34
phosphorus (Fe-bound)	(kg-P/t-ds)	5
carbon (biogenic)	(kg-C/t-ds)	325
carbon (non-biogenic)	(kg-C/t-ds)	32

Table 2: Metal and organic pollutant concentrations in the modelled biosolids stream

Aluminium	(g/kg-ds)	13.7	Aldrin	(µg/kg-ds)	10
Iron	(g/kg-ds)	5.5	Chlordane	(µg/kg-ds)	13
Arsenic	(mg/kg-ds)	6.4	DDD	(µg/kg-ds)	10
Cadmium	(mg/kg-ds)	1.1	DDE	(µg/kg-ds)	10
Chromium	(mg/kg-ds)	33	DDT	(µg/kg-ds)	12
Copper	(mg/kg-ds)	468	Dieldrin	(µg/kg-ds)	61
Lead	(mg/kg-ds)	19	Heptachlor	(µg/kg-ds)	12
Mercury	(mg/kg-ds)	0.8	Lindane	(µg/kg-ds)	10
Molybdenum	(mg/kg-ds)	6.8			
Nickel	(mg/kg-ds)	29			
Selenium	(mg/kg-ds)	4.1			
Zinc	(mg/kg-ds)	633			

Transport modelling

Transport distances for both disposal scenarios (40km for trucking to landfill; 200km for trucking to farms) were based on 2008 survey data collected for a large number of sewage treatment plants in South East Queensland, the broader urban region that includes the Gold Coast city (de Haas et al. 2009, Lane et al. under review). Models for the transport provision were taken from the Australasian LCA inventory database (Grant 2012), and are described in more detail in the supplementary information.

While a 200km distance to reach farmland might be excessive in the context of smaller centres in Australia, it is much less than the average 300km distance involved in carting biosolids in Australia's largest city (Peters and Rowley 2009). Similarly, distances to landfills (or stockpiles) could also vary substantially across different locations. The effect of using different assumptions for transport distance is considered in sensitivity analysis provided in the supplementary information.

Application to agricultural fields – on-farm handling

Tractor fuel use for the biosolids dispersal is taken from literature information (Foley et al. 2010). In the absence of data on the solids handling regimes used on Australian farm sites, the (possible) stockpiling of biosolids prior to application was not included in the modelling for this study. Given that such practice could potentially generate substantial quantities of CH_4 (Majumder et al. 2014), this issue warrants closer consideration in future analysis.

Application to agricultural fields – fertiliser displacement

The biosolids application rate is calculated according to the 1 NLBAR[1] criteria of the Queensland biosolids application guidelines (EPA 2002), and a default nitrogen demand of 456 kg-N/ha (per crop cycle) from the control site in the Barry and Bell (2006) trials.

Total availability of mineralised nitrogen is estimated assuming a 5 year biosolids application cycle. While that timeframe is often more frequent than would be justified on a strictly nutrient-limited basis, it has become commonplace in Australian biosolids management (Stevens et al. 2012). Our

[1] NLBAR = Nitrogen Limited Biosolids Application Rate

default scenario assumes that 65% of the biosolids P will be bioavailable (i.e. available for crop uptake) over the short-medium term, taken from data from specific trials under Australian conditions (Pritchard et al. 2010). This implies a phosphorus application ~15 times greater than the annual requirements (56 kg-P/ha) for the crop in question.

The modelling for this study assumes that 100% of the default crop N and P requirements are avoided in the year of biosolids application. However, no literature based information was found to guide an assumption on fertilisation regimes for years 2-5 of the application cycle. Given the complexity inherent in agri-system management, and the available evidence on phosphorus accumulation in Australian soils (e.g. McLaughlin et al. 2011), it would seem tenuous to assume that farmers would maintain strict adherence to the biosolids nutrient stoichiometry over time. For the sake of exploring this issue, we assume by default that our hypothetical farmer would utilise only 50% of the surplus biosolids N and P available for crop uptake in in years 2-5.

Manufacture of the (avoided) synthetic fertilisers, and transport of the product from manufacturing site to a regional store, were modelled using default inventories from the Australasian LCA inventory database (Grant 2012). We assumed that transport from the store to the farm would involve a distance of 50km, and that the application to soils would normally be incorporated into other field management activities (hence require no additional farm tractor movements).

Table 3: assumptions used for modelling of biosolids disposal scenarios

		application to farmland		**disposal to landfill**	
transport to site		200 km	survey data[1]	40 km	survey data[1]
onsite biosolids handling	power use	--	literature review of (Foley et al. 2010)	0.8 kWh/t-ws	Australasian Life-Cycle inventory database (Grant 2012)
	diesel use	0.325 L/t-ws		1.0 L/t-ws	
	application rate	17 t-ds/ha (nominally once every 4 years)	(Barry and Bell 2006)[2]	--	
fertiliser nutrients	total N avoided	456 kg-N/ha in the application year	as described in the supplementary info., based on data from (Barry and Bell 2006)[2]	n/a	
	total P avoided	56 kg-P/ha each year for 4 years			
N_2O & NH_3 emissions	NH_3 from biosolids	243 g-NH_3 per kg-N applied	literature review of (Foley et al. 2010)	61 g-NH_3 / kg-N applied	literature review of (Foley and Lant 2007)
	N_2O from biosolids	15.7 g-N_2O per kg-N applied	default emission factor from (IPCC 2006a)	6.9 g-N_2O / kg-N applied	
	N_2O from synthetic fertilisers	15.7 g-N_2O per kg-N applied		--	
carbon balance	non-biogenic carbon	9% of all biosolids carbon	methodology of (Lane et al. under review)	*as for application to farmland*	
	carbon sequestered	24% of carbon applied to soils	estimate from (Brown et al. 2010)	modelled according to Australian government reporting methodology (DCCEE 2011)	
	CH_4 from point of disposal	2.8 g-CH_4 per kg-ds applied	literature review of (Foley et al. 2010)		
	CH_4 from stockpiling	*excluded*		--	
	CH_4 recovered for power generation	--		31% of total CH_4 generated	overall recovery in Australian landfills in 2011 (DIICCSRTE 2013)
crop yield increase		330 kg/ha of sorghum grain in the year of application	(Barry and Bell 2006)[2]	n/a	

[1] *Taken from a 2008 survey of STPs in the Gold Coast & neighbouring regions. 200km is typical of the 9 plants that trucked their biosolids directly to farms. 40km is the average distance for the 5 plants that sent their biosolids to landfill. The survey formed part of the work presented in de Haas et al. (2009) – see also the supplementary information.*

[2] *Based on experimental data for the 2002/03 trials on sorghum crops at the Cecil Plains site, South-East Queensland, Australia (Barry and Bell 2006) – see also the supplementary information*

Application to agricultural fields – crop yield response

The 8% yield increase measured in the Cecil Plans sorghum trials (Barry and Bell 2006) is at the higher end of the spectrum in the available Australian field studies.

The crop offsets were calculated for an equivalent amount of grain production, using an unpublished inventory for feed-sorghum generated by agricultural LCA researchers in Australia (Renouf 2012). This captured all production inputs and outputs measured to the farm gate. Downstream processing operations were excluded, on the assumption that that these would be identical for both the displaced grain, and the grain produced on the biosolids-amended field.

Harvesting operations were included in the inventory for the displaced feed-sorghum production. However, we assumed that no additional harvesting would be required to capture the induced yield increase from the biosolids-amended field, on the premise that harvesting fuel inputs are determined by the area covered rather than by the quantity of produce extracted.

Application to agricultural fields – carbon & nitrogen fluxes

There is very little data available to guide quantitative estimates of nutrient flux to waterways from the agricultural field, at least in the Australian context. We therefore chose to investigate the implications of using one of the more accessible, default approaches that is available to LCA practitioners. To this end, flux rates of 5-6% (of applied N and P) were derived from the ReCiPe model for assessing *Eutrophication Potential* (Goedkoop et al. 2009, Struijs et al. 2011).

The soil carbon and nutrient balances were then completed by estimating gaseous fluxes of CH_4, CO_2, N_2O and NH_3. Overlaying the carbon balance is the assumption that 10% of the biosolids carbon is of fossil origin (as per Lane et al. under review), reflecting the high fraction of non-biogenic carbon detected in Australian sewage systems (Law et al. 2013). Other key assumptions are summarised in Table 3 and described more fully in Lane et al. (under review).

Disposal at the landfill

The modelling of landfill processing systems (power and fuel use; power generation) was based on the generic landfill inventories contained in the Australasian life-cycle inventory database of Life Cycle Strategies (Grant 2012).

Methane generation, and the overall landfill carbon balance were calculated with the 1st order decay model recommended for use in GHG accounting for Australian landfill sites (DCCEE 2011). The carbon balance used the same 10% estimate for non-biogenic carbon as described for the landfill disposal modelling, assuming that this non-biogenic fraction partitioned proportionately across all components of the soil carbon balance. N_2O generation from landfilled biosolids is also included in the analysis. Other relevant assumptions are summarised in Table 3, with more detailed description provided in the supplementary information.

Impact assessment modelling

The number of possible impact categories available through the LCA methodology is extensive. This study focuses on a subset of eight of those possibilities (Table 4). Each has some overlap with the scope of Australian research and/or debate on biosolids use for agriculture, as outlined above. Three (*AOD*, *TTX*, *GW*) align with the nominated priority categories for agricultural LCA development in Australia (Eady et al. 2012), while the other five are some of the most commonly used in the LCA domain.

Eutrophication

A single metric is used here for analysis of eutrophication hazards, expressed as an *Aquatic Oxygen Depletion (AOD)* score. This is deliberately intended to provide a more generic assessment than the regionally-specific approaches to eutrophication assessment favoured in many recent studies (e.g. Lane et al. under review).

To this end, the *AOD* metric accounts for implications in both marine and freshwater systems, and includes 100% of all N, P and COD releases into both those receiving environments.

Table 4: Impact assessment models considered for this study

Indicator	Scope of indicator assessment	Source of LCIA model
Aquatic Oxygen Depletion (AOD)	(Potential) de-oxygenation of freshwater & marine systems by nutrient (via algal growth) & organic matter inputs. Both direct (emission to waterways) & indirect (via airshed nitrogen) inputs are accounted for.	customised approach for generic assessment (see supplementary info)
Terrestrial Ecotoxicity (TTX)	(Potential) toxic impacts on ecosystems and humans caused, directly and indirectly, by emissions of metal & organic chemical species to air, land and waterways.	USES-LCAv2 *(van Zelm et al. 2009)*
Human Toxicity (HTP)		
Minerals Depletion (MD)	Depletion of (in ground) abiotic stocks, compared on the basis of the marginal cost of extraction (for *MD*) and the relative energy content (for *FFD*).	ReCiPe midpoint models (Goedkoop et al. 2009); + *MD* factor for phosphate deposits developed in this study
Fossil Fuels Depletion (FFD)		
Cumulative Energy Demand (CED)	Total primary energy footprint	Ecoinvent database (Frischknecht et al. 2007)
Global Warming (GW)	(Potential) for human and ecosystem impacts caused by climate change (*GW*) and by depletion of the stratospheric ozone layer (*OD*)	100yr GWP factors (IPCC 2007), with customisations for NH_3 and CH_4.
Ozone Depletion (OD)		steady state ODP factors for halocarbons (Daniel et al. 2011) & N_2O (Lane and Lant 2012).

Toxicity

USEtox is the preferred toxicity modelling package for many in the LCA research community (Hauschild et al. 2013, Rosenbaum et al. 2008), however does not provide a metric for assessing terrestrial ecotoxicity hazards. Since the onsite toxicity of biosolids application has been a high priority research topic in Australia (Pritchard et al. 2010), we have instead opted to use the USES-LCA toxicity model (van Zelm et al. 2009) for our analysis. USES-LCA provides metrics for both *Terrestrial Ecotoxicity (TTX)* and *Human Toxicity (HTX)*, was developed along very similar principles, and has been shown to correlate well with USEtox results (Hauschild et al. 2008).

Resource Depletion

A conventional approach for *Fossil Fuel Depletion (FFD)* was used, combining different fuel sources based on their relative energy content. This essentially provides a proxy for the overall life-cycle energy burden embedded in supply chains. *Cumulative Energy Demand (CED)* is also used to provide a more direct measure of overall primary energy use, this being one of the oldest metrics in the LCA toolkit.

Given the broad spectrum of metrics available for assessing *Minerals Depletion (MD)* in LCA, it seems likely that this component of resource depletion analysis in LCA will remain (to some degree at least) a matter of choosing the most appropriate metric for a particular analytical purpose (see the supplementary information). This is not, of itself, a concern, being entirely consistent with the procedural guidelines spelled out in ISO standards for conducting LCA (ISO 2006). The challenge for biosolids analysis, however, is that very few of the available metrics incorporate impact factors for the (avoided) depletion of mineral phosphorus resources.

In previous analysis of the same Gold Coast urban water system that is considered here (Lane et al. under review), the CML model for minerals resource assessment (van Oers et al. 2002) was chosen because it is one of the few to account for phosphate resources. The results in that study suggested that the substantial offset of mineral phosphorus fertilisers delivered only a negligible benefit in terms of resource sustainability. However, because of the different fundamentals utilised

by the different metrics that are available, further analysis is required to determine whether similar conclusions would be reached using different approaches to mineral resource assessment in LCA.

For the analysis in this study, we use a metric for minerals resource assessment taken from the ReCiPe modelling package (Goedkoop et al. 2009). Whereas the CML-based model is one of the simpler approaches available, being based on usage-stock ratios, the ReCiPe model uses a more complex metric that attempts to incorporate both changes in marginal ore quality, and stock scarcity, into the considerations. Further discussion on the merits of different approaches for assessing *Minerals Depletion* is provided in the supplementary information.

Since the published ReCiPe model does not consider phosphorous resources, we developed an interim impact factor for mineral phosphate rock deposits in a manner consistent with the ReCiPe methodology.

Greenhouse Gas emissions

Global Warming (*GW*) scores are calculated using 100yr GWP equivalency factors from the IPCC (2007), modified to include secondary N_2O caused indirectly by NH_3 gas emissions (IPCC 2006b), and characterisation factors for biogenic and fossil-sourced CH_4 emissions that include the secondary effects of oxidation to CO_2 (Muñoz et al. 2013). Soil sequestration of fossil fuel carbon is credited to the *GW* results.

Ozone Depletion (*OD*) results are also considered following the approach of Lane and Lant (2012), because of their sensitivity to anthropogenic N_2O emission loads.

RESULTS & DISCUSSION - UNCERTAINTIES IN THE MODELLING OF INVENTORY FLOWS FOR AGRICULTURAL BIOSOLIDS USE

GHG balances

The GHG implications of agricultural biosolids use are an issue that has received much attention in the literature. Historically, the majority of discussion has focussed on the opportunity to avoid energy-intensive fertiliser manufacture, at the expense of trucking a dilute biosolids stream over (potentially) long distances. However, the uncertainties associated with direct gaseous emissions from the farm paddock might be as, or more, important to the quality of GHG-focussed biosolids analysis.

Table 5 Greenhouse gas balance (expressed as Global Warming) for the agricultural-use biosolids pathway

		biosolids use	avoided fertiliser use	avoided crop production
field fluxes	N_2O	408	-220	-12
	NH_3	71	-19	-1
	CH_4 (biogenic & non-biogenic)	68		
	CO_2 (non-biogenic)	96		
	C sequestered (biogenic)	-274		
biosolids transport + application		153		
production + supply			-134	-10
other		4		
total		**527**	**-373**	**-24**
		130 kg CO2e / t-ds		

For the agricultural-use scenario, the net increase in N_2O generation (+176 kg-CO_2/t-ds) is similar in magnitude to the greenhouse penalty of the biosolids transport (+153 kg-CO_2e/t-ds), and the benefit (-134 kg-CO_2e/t-ds) from avoiding the need for synthetic fertiliser manufacture and supply (Table 5). The most obvious concern here is the use of a single emission factor based on total nitrogen application, despite the fact that biosolids nitrogen (predominantly organic) and the more conventional ammonium based products could have very different implications for soil N_2O biochemistry. To our knowledge, there is very little (if any) field study data on N_2O emissions from alternative fertilisers used under Australian conditions. Notwithstanding that uncertainty, the

concern that field N_2O balances could be an important element of biosolids analysis remains valid. N_2O generation rates in the available Australian field studies vary by two orders of magnitude, depending on the crop type, site location and prevailing management practices, with rates as high as 27% of applied N having been detected at some sites (Thorburn et al. 2013). Given that is an order of magnitude larger than the default emission factor used here (1% of applied N), it is clear that the issue of N_2O emissions warrants closer consideration in the biosolids debate.

Soil carbon balances might also be critical to the overall GHG intensity of agricultural biosolids use (Table 5). Even with our seemingly low estimate for CH_4 generation from the aerobic soil environment, this contribution to the overall GHG profile (68 kg-CO_2e/t-ds) was not trivial. So too, that attributed to soil mineralisation of non-biogenic carbon in the biosolids (96 kg-CO_2e/t-ds). The potential for soil carbon sequestration might be far more significant, showing here a *GW* reduction (274 kg-CO_2e/t-ds) that is the second largest influence across the system life-cycle. However, our sequestration factor (24% of carbon applied) is sourced from international field data, and its relevance for Australian conditions is questionable (see Lane et al. under review). More specifically for the case of biosolids use on farms, the limited available Australian field data is ambivalent on whether increases in soil carbon will be maintained beyond the short term (Barry and Bell 2006, Ives et al. 2011, Powell and Graham 2012). Until there is a more substantial body of evidence available, literature based assumptions (such as the one used here) may be valid only for exploratory analysis.

With such potential for variation in positive and negative GHG sources, across so many different parameters, it would seem unlikely that generalised conclusions could be reached on the net GHG implications of adopting the practice to apply biosolids directly to farms as a fertiliser substitute. More robust quantitative uncertainty analysis across all the possible factors would be beneficial, but was beyond the scope of this study. Given the paucity of synthesis information available in the literature, this is also likely to be beyond the reach of most future LCA of Australian agricultural outputs. This is concerning, as it suggests there is great potential for future studies to inappropriately generalise the assumptions used for LCA critique of agricultural biosolids use in different Australian case studies

Nutrient losses to waterways

The large increase in overall *Aquatic Oxygen Depletion (AOD)* results (Table 9; Table 10) illustrates the problematic implications of using the default loss rates embedded in the ReCiPe model for eutrophication. The available Australian research does highlight the potential for nitrate and phosphate leaching from biosolids application to be higher than from the use of conventional fertilisers, even at 1 NLBAR application rates (Ives et al. 2011, Pritchard et al. 2007, Pu et al. 2008). However there is no evidence to suggest this would eventuate if biosolids applications were managed appropriately, nor any to justify the large loss rates built in to the ReCiPe model. The latter rely on European based assumptions for leaching fluxes, which are generally considered to be much higher than the equivalent losses in Australia.

For analysis of biosolids disposal options in the context of Australian farming practices, it would be preferable to use more focussed information for estimating nutrient losses from different fertiliser types. This would also need to account for the influence of varying soil type, climate and farm management practices. Unfortunately, to our knowledge, there is little published material available to guide such an assessment in a standardised manner.

In the absence of locally sensitive information on nutrient flux estimates to guide generic LCA analysis of Australian agriculture, case-study specific soil-nutrient modelling might be required for robust LCA critique of the biosolids-to-agriculture option.

Estimating the offsets

Despite the enthusiasm in Australia for the potential fertiliser offsets and crop yield changes associated with biosolids use on farms, there are gaps in the publicly available knowledgebase that impede the inclusion of these issues into the quantitative, environmental LCA framework. Because

of this, sensitivity analysis was undertaken to explore two sets of variations to the default assumptions used for this study (Table 6).

The first test recognises that, over the long term, the actual reduction in use of synthetic phosphorus fertilisers could be much lower, or higher, than our default estimate. This was achieved by varying the fraction of surplus bioavailable phosphorus that displaces conventional sources, spanning the two possible extremes: (i) the farmer reverts to conventional phosphorus application in years 2 to 5, meaning the overall offset is limited to the year 1 benefits (56 kg-P/ha); and (ii) the use of mineral phosphate fertiliser is avoided completely for the duration of the 5 year biosolids application cycle.

In between these two extremes might be the situation where farmers juggle over time their biosolids application regime to meet a range of competing objectives. This seems (to us) a plausible outcome, but represents a somewhat difficult situation to translate into quantitative predictions.

The second test considers a single range of crop offset possibilities, but essentially covering a multitude of uncertain factors. Firstly, there is the question of whether or not the short term crop yield changes measured in Australian studies would be maintained over the long term. A further challenge exists in determining how that crop yield change will influence the commodity marketplace, since it is this influence that should be modelled when quantifying the implications of any change in crop yield on the biosolids-amended field. While the LCA research community has defined principles for determining the marginal supplier of goods and services in cases such as this, there is a distinct lack of such data in Australia that could guide the LCA analyst. The range considered here spans from:

- a zero offset, representing either (a) no yield benefit being achieved; or (b) no displacement of production is actually achieved;
- an offset that is (up to) double our default assumption, representing either (a) a doubling of the overall yield benefit (whether in year 1, or spread over intermediate years between biosolids applications); or (b) the displacement of a crop production system that is twice as environmentally intensive as our chosen process.

Table 6: Impact results attributed to biosolids disposal under the agriculture-use scenario (from Table 9b) expressed per unit mass of dry solids. Also shown is the variation in those results caused by changing (±100%) the default assumptions for modelling the displacement of synthetic fertilisers, and the increase in crop yield that is attributed to the biosolids application

	total impacts of the agri-use disposal pathway (using the default parameterisation)		% change to the default result, by changing assumptions for…	
			P mgmt practices (0→100% of 5yr-surplus bioavailable P is utilised	crop offsets (0→200% of default crop offsets are realised)
AOD	231	kg-O2-e/t-ds	± 35%	± 2%
TTX	4	kg-1,4-DCB-e/t-ds	± 1%	± 0%
HTX	342	kg-1,4-DCB-e/t-ds	± 37%	± 2%
MD	-172	kg-Fe-e/t-ds	± 75%	± 2%
FFD	-4	kg-oil-e/t-ds	± 203%	± 81%
CED	-216	MJ/t-ds	± 178%	± 71%
GW	130	kg-CO2-e/t-ds	± 21%	± 16%
OD	14	g-CFC11-e/t-ds	± 1%	± 6%

Varying the offset parameters has a notable effect on most of the impact category results associated with the biosolids disposal component of the urban water system (Table 6). The uncertainty involved in estimating these fertiliser and crop offsets could therefore have an important influence on biosolids focussed analysis using the LCA methodology.

The sensitivity results also reinforce the conclusion (see also Table 9, Table 5) that, despite the deliberate use of field data with a relatively large crop yield response, the displaced fertiliser use

delivers greater benefits to the LCA results than do the crop offsets. That conclusion contrasts somewhat with the financial analysis undertaken during the field trial study (Barry and Bell 2006), which indicated similar benefits associated with each of the two factors.

RESULTS & DISCUSSION - SHORTCOMINGS IN THE AVAILABLE LIFE-CYCLE IMPACT ASSESSMENT MODELS

Toxicity impacts

The biggest increases in the life-cycle impacts associated with change in disposal pathways are for the categories of *Terrestrial-* and *Human- Toxicity* (Table 10). However, of those two, only the change in the ecotoxicity results appears significant when benchmarked against the life-cycle environmental burden of the cropping system. The remainder of this discussion therefore focusses on the ecotoxicity category.

Trace metals in the biosolids, particularly Copper (Cu), Zinc (Zn) and Mercury (Hg), are the dominant cause of the *Terrestrial Ecotoxicity* results for the scenario with biosolids use on agricultural soils (Table 7). The toxicity offsets from avoiding heavy metals in diammonium phosphate (DAP) were two orders of magnitude lower than the impacts ascribed to the biosolids. In part this can be attributed to the high copper (468 mg/kg-ds) and zinc (633 mg/kg-ds) concentrations in our hypothetical biosolids assay, and the much higher ratio of metals to nutrients in the biosolids. Substitution of DAP with biosolids on the basis of phosphorus content will therefore lead to an overall increase in soil metal loadings (Foley et al. 2010).

Table 7: Terrestrial Ecotoxicity results for the agricultural-use scenario, broken down by toxicant group and system component. Contributions less than 0.1% are not shown. Convention in LCA toxicity assessment is to treat contributions <1% as statistically insignificant (Rosenbaum et al. 2008).

		Cu	Zn	Hg	Se	Cd	other metals	organics	other
biosolids application	biosolids contaminants	65%	17%	5%	3%	0.1%	1%	0.1%	
	transport & application								
displaced fertiliser	fertiliser contaminants		-0.3%			-0.2%	-0.1%		
	manufacture & supply						-0.15%		
displaced crop production	fertiliser contaminants								
	other								
remainder of the urban water system	operations	0.7%	0.4%	0.3%			1.5%	4.8%	
	construction	0.1%	0.2%	0.3%			0.2%		
total		44.3 t 1,4-DCB-e/y (95%)						2.3 t 1,4-DCB-e/y (5%)	

The dominance of the biosolids' metals to the ecotoxicity results contradicts two relatively high profile scientific maxims. Firstly, it does not match the conclusions from recent Australian research that the copper and zinc in biosolids pose little toxicological threat to Australian soils if applied at typical rates (Pritchard et al. 2010). Secondly, it is inconsistent with the argument that alternative phosphorus sources are needed to avoid the future use of synthetic fertilisers with trace concentration of heavy metals - concentrations that are expected to increase over time as lower grade phosphate-ore bodies are utilised (Cordell et al. 2009).

It is clear that LCA *Terrestrial Ecotoxicity* models are not yet capable of providing useful guidance when it comes to biosolids management, and should be excluded from analysis of biosolids disposal options. Of itself, this is not a novel conclusion – this constraint has been recognised for some years by academic and industry analysts using LCA to focus on biosolids disposal options (e.g. Corominas et al. 2013, Hospido et al. 2005, Peters and Rowley 2009). However it appears that message has had less influence than might be expected, with water industry studies

continuing to highlight the "significance" of biosolids metals using LCA toxicity models (e.g. Mahgoub et al. 2010, Pasqualino et al. 2009).

While it remains valid to exclude LCA toxicity impacts from biosolids analysis aimed at providing insight to the urban water industry, this may not be the case if the primary motivation for conducting LCA of biosolids use becomes the agricultural perspective. Toxicity analysis has been identified as a priority for inclusion in the expansion of LCA applied to Australian agriculture (Eady et al. 2012), reflecting the desire to incorporate agri-specific issues such as pesticides use and synthetic fertiliser contaminants, and benchmark such results against the potential impacts that occur elsewhere in the system lifecycle.

Minerals resource depletion

The avoided *Minerals Depletion* is the biggest benefit identified in the change of biosolids disposal pathways (see Table 9; Table 10), resulting from the reduced demand for synthetic fertilisers produced from mined phosphorus supplies. This contrasts strongly with the conclusions drawn from previous analysis using the CML-based metric (Lane et al. under review), where the biosolids phosphorus recovery showed a negligible benefit. The contrasting conclusions cannot be attributed to differences in inventory level calculations – the generation of biosolids phosphorus is modelled identically, and the effective P displacement (25%) in this study is much lower than the offset ratio (50%) used in the previous work.

However, for a direct comparison with the results from that previous study, the analysis needs to also account for the minerals extraction associated with the construction stage of the urban water system life-cycle (including those minerals literally embedded in the infrastructure stock). With the system boundary extended in that manner, the results using the ReCiPe based metric do not unequivocally support the notion that biosolids reuse is the best mechanism by which the water industry can help alleviate global minerals resource sustainability concerns (Table 8). Depending on the degree of P offsets that actually get delivered over the long term, the *Minerals Depletion* incurred through the infrastructure construction, and even through the use of chemicals for water and wastewater treatment, may be of a similar order to that avoided by the biosolids nutrient recovery. While not considered here, such a comparison would also be sensitive to the choice of chemicals used, or the assumptions made about infrastructure replacement schedules.

As in that previous study (Lane et al. under review), the analysis provided here again implies that the case for biosolids nutrient recovery, if justified on the grounds of resource sustainability, requires further critique. According to the LCA analytical frameworks that have been employed, similar 'sustainability' benefits might be achieved by reducing treatment plant chemicals use, designing infrastructure systems that are less materially intensive, or by choosing different materials for the construction of treatment plant infrastructure.

Table 8: Minerals Depletion results for the agricultural-use scenario, showing the overall variation in these results using the range of fertiliser displacement assumptions applied in the previous sensitivity analysis (see Table 6). The central column provides the results using the default assumptions (as per Table 9b), adjusted to also include the contributions attributed to construction of the urban water system infrastructure.

		Minerals Depletion (t Fe-e/y)		
		low P displacement	default P displacement	high P displacement
displaced fertiliser - biosolids use on farms		**-458**	**-1833**	**-3273**
total for rest of system		+1142	+1142	+1142
displaced fertiliser - urban WW irrigation			*-143*	
operational inputs for the urban water system	*chemicals*		*+153*	
	other		*+38*	
construction of the urban water system infrastructure	*networks*		*+39*	
	other		*+1055*	
urban water system - overall		**+684**	**-691**	**-2131**

This critique should involve further review of the way that phosphate rock resources are handled under LCA metrics. Neither of the *Minerals Depletion* metrics employed in the two studies discussed here can be considered a definitive choice for urban water systems analysis (see the Supplementary information). Recent reviews have argued the need for, and proposed, new approaches for assessing this issue in LCA (Swart and Dewulf 2013, Vieira et al. 2011, Vieira et al. 2012, Yellishetty et al. 2011). Given the large variation in the significance ascribed to phosphorus depletion across the different metrics reviewed here (Figure s2), it is not possible to predict whether our conclusions would hold using a broader spectrum of the models that are available. More fundamentally, consideration should be given to whether LCA resource depletion metrics can adequately represent the somewhat unique socio-economic characteristics of the phosphorus resource sustainability challenge (see the supplementary information).

RESULTS & DISCUSSION - LCA PERSPECTIVES ON CHANGING THE METHOD OF BIOSOLIDS DISPOSAL

This section introduces two additional aspects to the analysis in this paper. Firstly, the implications of biosolids disposal in landfill are considered, so as to provide a benchmark as required for quantifying the full life-cycle implications of biosolids use on farms. Secondly, the impacts associated with the biosolids management options are benchmarked against those associated with (a) an overall urban water system; and (b) an overall crop production system.

For the landfill scenario, the biosolids disposal process makes a number of small but notable contributions to the overall life-cycle impacts of the urban water system (Table 9a). Direct emissions of CH_4 and N_2O from the landfill site are the primary contributors to the *Global Warming (GW)* and *Ozone Depletion (OD)* impacts respectively. Landfill carbon sequestration reduces the *GW* results by a third. The credits associated with power generation at the landfill site are much smaller, as is the GHG footprint of the solids transport from the STP to the landfill site. The solids transport is, however, the primary source of toxicity impacts associated with the biosolids disposal.

With the biosolids applied to farm soils, most of the impact category results increase or decrease dramatically (Table 9). Taken at face value, the LCA results suggest a substantial increase in the potential for nutrient discharge to nearby receiving waters (reflected in the increased *Aquatic Oxygen Depletion* result), and a substantial increase in the potential for exposure to toxic chemicals (reflected in the increased results for *Terrestrial Ecotoxicity* and *Human Toxicity*). As noted above, closer scrutiny of approaches used to generate those results would be required, in order to reach an understanding on the relevance of these tradeoffs.

Another observation from comparing the two methods is that the net balances for energy use (reflected in the CED and *FFD*) and GHG emissions (as *GW*) are as or more sensitive to the amount of CH_4 recovered at the default landfill, than they are to the biosolids transport distances involved (supplementary info; also Table 9). This highlights the importance of factoring in the status quo disposal option, when ascertaining the true energy implications of a decision to apply biosolids to agricultural soil.

The scale of these changes is substantial, regardless of whether the system boundary is set at the wastewater system, or more broadly includes the entire urban water system. Some appear even more significant if benchmarked against the estimate for overall life-cycle impacts of the site farming system (Table 10). This is not surprising for the *AOD* and *GW* results; given it is other components of the urban water system that are the major contributors to its overall impacts in those categories. The toxicity impact increases appear much less substantial when benchmarked against the cropping system, but are still very large (+496%) for the terrestrial ecotoxicity (*TTX*) results. In a similar vein, the displacement of synthetic fertiliser use reduces (by 84%) the *Minerals Depletion* associated with the cropping system, although this is a much less dramatic change than for the urban water system.

Table 9: Selected LCA impact results for the operations of a city-scale urban water system, assuming that all biosolids are (a) sent to landfill; (b) applied directly to farms. The modelling of "WW system – overall" and "Urban water system" includes the biosolids disposal, and is described further in Section B of the supplementary information provided in Appendix C2.

(a)

	AOD (kt O_2-e/y)	TTX (t 1,4-DCB-e/y)	HTX (t 1,4-DCB-e/y)	MD (t Fe-e/y)	FFD (t oil-e/y)	CED (TJ)	GW (kt CO2-e/y)	OD (kg CFC11-e/y)
Biosolids - overall	**0.11**	**0.03**	**0.49**	**-2**	**-164**	**-13**	**15.4**	**105**
transport to disposal site	0.002	0.01	0.30		189	7.921	0.5	0.17
onsite - energy use			0.001	0.06	11	0.657	0.05	0.05
onsite - gas emissions	0.12	0.02	0.21		76	3.209	23	107
onsite - C sequestration							-6.4	
offsets – power generation	-0.006	-0.01	-0.03	-2	-440	-25	-1.8	-2.0
WW system - overall *(% contribution from biosolids disposal)*	**15** 1%	**3** 1%	**2.0** 24%	**-77** 3%	**11,944** -1%	**673** -2%	**78** 20%	**512** 20%
Urban water system - overall *(% contribution from biosolids disposal)*	**15** 1%	**4** 1%	**3.2** 15%	**82** -3%	**17,277** -1%	**950** -1%	**103** 15%	**532** 20%

(b)

	AOD (kt O_2-e/y)	TTX (t 1,4-DCB-e/y)	HTX (t 1,4-DCB-e/y)	MD (t Fe-e/y)	FFD (t oil-e/y)	CED (TJ)	GW (kt CO2-e/y)	OD (kg CFC11-e/y)
Biosolids - overall	**2.5**	**42.6**	**3.7**	**-1,868**	**-49**	**-2**	**1.4**	**147**
transport to disposal site	0.01	0.02	0.9		547	23	1.6	0.5
onsite - energy use			0.04		25	1	0.1	0.04
onsite - gas emissions	0.3		-0.004		-3	-0.1	4.4	156
onsite - C sequestration							-3.0	
onsite - other	2.2	43.0	3.3				0.03	2.0
offsets - fertiliser supply to site	-0.01	-0.09	-0.5	-1,867	-612	-26	-1.5	-3.0
offsets - crop production	-0.01	-0.001	-0.01	-1	-6	-0.3	-0.2	-8.2
WW system - overall *(% contribution from biosolids disposal)*	**18** 14%	**46** 93%	**5.2** 71%	**-1,942** 96%	**12,059** -0.4%	**684** -0.3%	**64** 2.2%	**554** 27%
Urban water system - overall *(% contribution from biosolids disposal)*	**18** 14%	**47** 92%	**6.4** 58%	**-1,784** 105%	**17,392** -0.3%	**961** -0.2%	**89** 1.6%	**574** 26%

Table 10: Change in the overall life-cycle impacts for three benchmark systems, associated with the decision to apply biosolids to agricultural fields. The change measured is the difference in impacts allocated to biosolids disposal for the two pathways presented in Table 9. The 'WW management system' and 'Urban Water System' benchmarks are those modelled for Table 9a. The 'crop production' benchmark is described further in Supplementary Information.

	AOD (kt O_2-e/y)	TTX (t 1,4-DCB-e/y)	HTX (t 1,4-DCB-e/y)	MD (t Fe-e/y)	FFD (t oil-e/y)	CED (TJ)	GW (kt CO2-e/y)	OD (kg CFC11-e/y)
change in total impacts of biosolids disposal	**2.4**	**43**	**3.2**	**-1866**	**115**	**11**	**-14**	**42**
change in total impacts for operating the wastewater system	+16%	+1,231%	+162%	-2,439%	+1.0%	+1.6%	-18%	+8%
change in total impacts for operating the urban water system	+16%	+1,157%	+101%	-2,278%	+0.7%	+1.2%	-14%	+8%
change in total impacts of crop production (5 yr cycle)	+65%	+496%	+10%	-84%	+5%	+11%	-97%	+8%

Taken at face value, these LCA results would suggest there are substantial downsides, along with the upsides, to the use of biosolids as an agricultural supplement. This conclusion applies regardless of whether such a decision is considered from the perspective of the water or agricultural industries.

The concern for the water industry is that, as the use of LCA for assessing biosolids use on Australian farms grows, there will be more and more occasions where such conclusions are reached without paying sufficient heed to the fidelity of the results.

CONCLUSIONS

Our results indicate that LCA applied to Australian agricultural systems using STP biosolids could misrepresent the significance of a number of important issues. The concerns can be grouped in one of two ways.

Firstly, there are important gaps in the knowledge base required for robust LCA, in part because certain key issues do not currently feature as a priority in Australian research and policy debate on the use of biosolids as a farming supplement.

- The historical focus on energy tradeoffs between biosolids transport and nitrogenous fertiliser manufacture is unlikely to adequately represent the scale or profile of greenhouse gas emissions associated with biosolids use on farms. A diverse set of possible field fluxes (C, CO_2, CH_4, N_2O) could be as, or more, important. Critically, the uncertainty associated with all these fluxes is very large, and many are likely to vary substantially across location, crop type and field management practices. More detailed investigation across the full spectrum of possible GHG contributors is required, before choosing which specific issues should be the focus of future research efforts.
- Eutrophication assessment has a prominent place in the LCA research domain, and the risk of biosolids nutrient flux to waterways has been identified by previous Australian research. However, there is a lack of informative science that would help translate those concerns into quantitative assumptions as required for use in LCA studies. Furthermore, the default approaches that are readily accessible to LCA analysts may be inaccurate in the Australian context. For robust incorporation of this issue into LCA-based analysis of biosolids use on farms, resource intensive modelling of nutrient fluxes may be required on a case by case basis.
- There is a tendency in LCA studies to estimate key field-based nutrient fluxes based on total nutrient application rates. This may not be appropriate for biosolids analysis, since biosolids application will typically result in greater total N and P loading rates than if using conventional synthetic fertilisers. The use of leaching and N_2O emission rates scaled to TN/TP application could bias against the biosolids option, overlooking any changes in local field management practices to compensate for the use of the different nutrient source.
- There are also large uncertainties in incorporating fertiliser and cropping production offsets into the analysis, which could have an important influence on LCA analysis focussed on the biosolids and cropping systems. The problem is not with the LCA methodology per se, which is well suited to including complex supply-chain interactions into the analytical scope, but with the lack of data available for quantifying these implications over the longer term. This challenge is compounded by the knowledge gaps being more to do with socio-economic aspects of the agricultural system, than biophysical attributes that are often better understood by LCA practitioners.

The second group of challenges relates to the fidelity of impact assessment models frequently employed in LCA research. Some of those most relevant to the biosolids disposal debate may be inadequate for the task.

- When compared to the risk analysis delivered by numerous Australian research studies, readily accessible LCIA models for terrestrial ecotoxicity and human toxicity substantially overstate the significance of biosolids metals. These models should be excluded from LCA applied to urban

water systems analysis, rather than used to selectively assess supply chain contributions while discounting the influence of biosolids contaminants.

- A novel approach to assessing phosphorus recovery provides very different results to off-the-shelf LCA based methods, but only weak support for the notion that biosolids use on farms should be justified on the grounds of mineral resource sustainability. However, the available metrics that do account for phosphorus resources provide only limited perspectives, and it is not possible to ascertain whether such conclusions might hold using the diverse set of alternative approaches that have been proposed in recent years.

In conclusion, it seems unlikely that LCA analysis of biosolids use for agriculture can give meaningful results that are representative of industry priorities and scientific knowledge, and allow for robust conclusions to be reached.

While many of these discrepancies have been recognised for some years, they have either been overlooked or excluded in most LCA analysis to date. This has been possible while the primary focus of such studies has been the needs of the urban water industry - identifying the best (or least bad) disposal option for urban water utilities, and understanding in particular the engineering based components of the biosolids disposal system. However this analytical approach only remains valid if it is the water industry that continues to drive the decision making on whether biosolids is used as a farming supplement.

If the decision on whether or not to use biosolids on farmlands becomes influenced more by the agricultural perspective, then it becomes less likely that such uncertainties can be avoided. Analysis to support a decision on which type of fertiliser to use might require a very different set of methodological approaches (system boundaries, inventory assumptions, impact categories), some of which could make biosolids reuse look substantially less favourable than is indicated by current scientific wisdom. There is the distinct possibility that LCA use by the agricultural industry could introduce a bias against the direct application of biosolids to farmlands.

The use of LCA to inform Australian agricultural management decisions is set to grow, driven by the international push for greater information on the environmental attributes of different products being traded in the global marketplace. If this were to reduce enthusiasm in the agricultural industry for utilising biosolids as a crop supplement, this could pose a substantial risk over the medium to longer term for those Australian urban water utilities already heavily reliant on this agricultural disposal option.

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APPENDIX C2 CASE STUDY #2 – BIOSOLIDS AND AGRICULTURE – A GROWING 'LCA RISK' FOR URBAN WATER UTILITIES

– SUPPLEMENTARY INFORMATION

This manuscript is in preparation as:

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BIOSOLIDS AND AGRICULTURE - A GROWING 'LCA RISK' FOR URBAN WATER UTILITIES

- SUPPLEMENTARY INFORMATION -

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A. Landfill and transport modelling

Landfill greenhouse gas emissions

The landfill carbon balance was based on the 1st order decay model recommended for use in greenhouse gas (GHG) accounting for Australian landfill sites (DCCEE 2011), producing estimates for carbon stored in the landfill, and fluxes of CH_4 and CO_2 generated at the site. Other information sources were used to complete the final balance of emissions for the landfill disposal modelling. The calculation procedure and underpinning assumptions are summarised as follows:

- 50% of the biosolids carbon is assumed to degrade over the long term, with 50% of that degraded fraction being converted to CH_4 (DCCEE 2011). We assume that the non-degraded fraction is stored (sequestered) in the landfill biomass over the long term.
- As a default, we assumed that 31% of the landfill-generated CH_4 is captured and diverted to a power generation facility, being the weighted average of overall CH_4 recovery in Australian landfills for the year 2011 (DIICCSRTE 2013). The emissions profile for the gas combustion was taken from the Australasian LCI database (Grant 2012). The power generated is fed into the state's high voltage grid, effectively displacing the need for the same power source mix as used for inputs to the landfill and other components of the local urban water system model (described in Lane *et al.* under review).
- 10% of the unrecovered CH_4 fraction is oxidised by aerobic bacterial activity in the cover layers of the landfill mass, following the standard approach in DCCEE (2011).
- Overall CO_2 flux from the landfill is calculated to complete the carbon balance, net of the sequestered component and CH_4 removed from the system.
- Overlaying that carbon balance is the assumption that 10% of biosolids carbon is of fossil (non-biogenic) origin, following the approach described in Lane *et al.* (under review). While this possibility is not commonly considered in wastewater system GHG analysis, it reflects empirical data on the high fraction of non-biogenic carbon in sewage inflows to treatment plants in neighbouring regions to the Gold Coast city (Law *et al.* 2013). It was assumed that the non-biogenic fraction of the biosolids carbon would partition proportionately across all carbon fluxes in the landfill system. In other words, all carbon fluxes (CO_2, CH_4, C sequestered) were assumed to have that same 10% non-biogenic fraction.

Literature studies provide a wide range of values for estimating N_2O emissions from biosolids contained in landfill. The reasons for this variation are not clear, and it may be they reflect regional differences in landfill management practices. For example, the higher-end recommendations (24 g-N_2O per kg-N applied to the soil) of Brown *et al.* (2010) would seem inconsistent with the expectation that N_2O emission rates from well-managed, anaerobic landfill processes would be extremely low (e.g. Foley *et al.* 2007). Reasons behind that difference of opinion were not explored, although may be associated with assumptions about landfill management practices. The potential for N_2O generation could be elevated substantially if the nitrogen rich biosolids are used as the cover material, rather than buried in the bulk landfill mass (Borjesson *et al.* 1997; Bogner *et al.* 2011). For this study, we assumed the latter practice would be in place, and used an emission factor (6.9 g-N_2O per kg-N in the biosolids) at the lower end of the spectrum. The chosen value is taken from the median of literature values for landfill disposal collated by Foley et al (2007).

Transport modelling

Transport of the biosolids was modelled as being by articulated tanker, allowing for no backhaul of secondary material.

Sensitivity analysis on landfill & transport assumptions

To explore the sensitivity of the energy and GHG balances to non-agricultural issues, three sets of variations were considered for the landfill and transport assumptions:

1. Variation in the landfill CH_4 recovery for power generation (LFG)

The default recovery rate (31%) may have little relevance for particular regional case studies in Australia, being deliberately chosen to represent an average of the 'overall' situation in Australia. This is also the default value used in some Australian LCI databases, and the approach used here therefore illustrates the implications of utilising such databases without undertaking a critical review of their relevance for any specific case study. Note, however, that the chosen is not dissimilar to the empirically-based estimates (41% recovery) for a system with a poorly performing landfill cover (Spokas *et al.* 2006), or used as the default assumption (30%) in a more recently developed model for developing landfill CH_4 inventory estimates (Spokas *et al.* 2011).

For the sensitivity analysis, the fraction of recovered methane was varied from 0→94%, reflecting the fact this will vary greatly depending on the location. The lower end estimate (0%) acknowledges that CH_4 recovery is not commonplace at smaller landfill sites in Australia, occurring mostly in the bigger urban centres. The upper end limit is taken from the estimated recovery rate for a retired landfill cell that has been capped with a clay liner (Spokas et al. 2006).

2. Variation in the landfill CH_4 recovery for flaring (LFF)

This uses the same set of CH_4 recovery assumptions as for *LFG*, but this time assuming the CH_4 is flared rather than used for electricity generation.

3. Variation in the net transport distances (T)

For the sensitivity analysis, the net transport differential between the two scenarios was varied from 0km (i.e. both options require the same transport distance) to 300km (i.e. the distance from STP to farm is 300km further than from STP to landfill). The latter represents a typical distance for biosolids transport from STP to farm in Australia's largest city (Lundie *et al.* 2004).

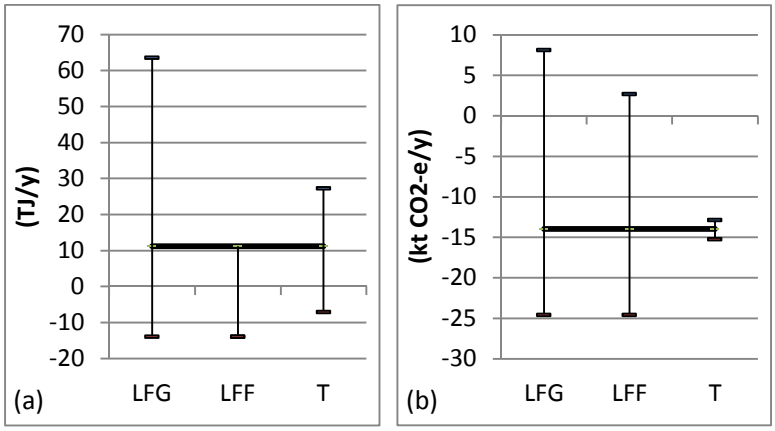

Figure s1: Sensitivity of the (a) CED and (b) GW balances to varying the assumptions for: (i) fraction of landfill generated CH_4 that is captured and used for power generation (LFG); (ii) fraction of landfill generated CH_4 that is captured, but assuming that all captured gas is flared (LFF); and (iii) net difference in transport distance between the landfill and agricultural-use disposal scenarios (T). The heavy black lines represent the net change between the two scenarios (+10 TJ/y for CED; -14 kt-CO_2e/y for GW) using the default assumptions.

The potential variability in both factors is large enough that the net energy balance for the change in biosolids disposal options could be positive or negative (Figure s1). However, it is the capture

and use of landfill CH_4 that offers greater potential for energy benefits. The landfill CH_4 balance also has a much stronger influence on the net GHG footprint. Once again, the potential variation depending on different landfill site practices is enough to dictate whether the biosolids management change delivers a net reduction or increase in overall GHG footprint.

Given the extreme sensitivity of the GHG results to variation in landfill CH_4 recovery practices, future analysis should also consider the uncertainties associated with estimating CH_4 oxidation and N_2O generation in landfill cover layers. The default oxidation rate used here (10%) is consistent with guidelines produced both the Australian government (DCCEE 2011) and IPCC (IPCC 2006), but there are a number of studies available indicating this could be a substantial underestimate of the actual oxidation that occurs at many landfill sites (e.g. Spokas *et al.* 2006; Bogner *et al.* 2011; Spokas *et al.* 2011). Notably, this oxidation rate has been found to vary strongly with the degree of subsurface CH_4 diversion that is achieved (Bogner *et al.* 2011; Spokas *et al.* 2011). As discussed above, N_2O flux rates attributable to biosolids disposal in landfills could depend strongly on the management practices in place. Given both processes will be sensitive to temporal factors (seasonal changes; changes in cover status), it may be that future biosolids LCA might benefit from directly modelling these temporal dynamics, rather than (as done here) assuming that overall partitioning factors can adequately represent the life-cycle GHG balances involved.

In conclusion, the sensitivity analysis indicates that the true energy and greenhouse gas implications of choosing to use biosolids on farms could be far more susceptible to the choice of baseline biosolids disposal method, than to the distances from STP to farm. This contradicts somewhat the notion that transport distance is a key design consideration (and uncertainty) in modelling the sustainability benefits of different biosolids disposal options (Peters *et al.* 2009).

More generally, the results also reinforce the notion that, given the prominence of fugitive CH_4 and N_2O emissions from so many parts of the urban water system, energy use will make a poor proxy for understanding the GHG implications of many decisions facing urban water managers (Lane *et al.* under review).

B. Urban water system benchmark

Overall wastewater management system

The benchmark for the overall wastewater management system is based on data collected in a previous study (Lane *et al.* under review), covering the same four STPs that provide the biosolids generation data used in this study. The benchmark model includes sewage collection and treatment, discharge and recycling of the treated wastewater, and disposal of screenings and biosolids.

The only change for the benchmark model was to assume that all the biosolids are sent to landfill. This then provides the baseline model for the change in disposal practices (from landfill to agricultural use) that is considered here.

This revised system definition was used to recalculate the benchmark LCA results, using the impact assessment models specifically chosen for this study.

Overall urban water system

The benchmark used for the overall urban water system combines the wastewater system (above) with the water supply system modelled in the 'Traditional infrastructure' scenario of (Lane *et al.* under review). The latter is a relatively low-energy use system where the majority of demand (84%) is met from local freshwater dams. Recycling of low-grade wastewater and, and household scale rainwater tanks, comprise the other local sources.

As described for the wastewater management system above, the life-cycle impact assessment (LCIA) results for this benchmark were recalculated using the impact assessment models chosen for this study.

C. Modelling the cropping system

The 195,000 t-ds/y of biosolids applied directly to farmland in Australia are used for a variety of different broadacre cropping systems, on a range of different soil types, across the country.

The cropping system used in this study is modelled after 2002/03 field trial data on a sorghum grain crop grown near the township of Cecil Plains on the Darling Downs in South East Queensland (Barry *et al.* 2006). From that study, data was only taken for trials with biosolids applied at 1 NLBAR, so as to match the underlying premise of our system boundary. This example was chosen because (a) it represents one of the larger (positive) crop yield responses that have been detected in Australian trials from biosolids applied at 1 NLBAR; and (b) the data is repeated for both aerobically-digested and anaerobically-digested biosolids. Where appropriate, assumptions used for this study are taken as the average of the field data reported for those two different biosolids streams.

The overall biosolids application rate for this case study was assumed to be 17t-ds/ha, matching the average of those used in the Cecil Plains trials.

To create a cropping system benchmark inventory that aligns with the production of 1 years' worth of biosolids, 5 years of repeated grain production is modelled so as to match the duration of the assumed biosolids application cycle. The overall grain production for this benchmark system was then calculated based on the measured grain production (4.12 t-grain/ha) from the 'control' field in the Cecil Plains trials, as follows:

total crop area (ha) included in benchmark
= total annual biosolids produced [10.8 kt-ds] ÷ biosolids application rate [17t-ds/ha]
= 638 ha

total benchmark production = 638 ha × 4.12 t- grain/ha × 5 crop cycles
= 13 kt-grain

D. Irrigation offsets

Biosolids that are applied directly to agricultural fields typically have high moisture content (~85%); however we have assumed that this would not affect the volume of irrigation water taken from local streams. Previous studies have raised the prospect that biosolids application could deliver substantial reductions in stream water use for irrigation-intensive crops such as rice (Peters et al. 2009). However there is very little published data on this issue in the available Australian research literature, making it difficult to gauge the likelihood of this occurring. Furthermore, direct benefits would only accrue where irrigation systems are actually in place. Where crop irrigation doesn't occur, any benefits to the crop from the biosolids water might instead translate into crop yield improvements, and therefore be addressed through our incorporation of crop offsets.

E. Assessing the *Minerals Depletion*

A diverse range of models exist for assessing the sustainability implications of resource depletion (or recovery) in LCA, without any consensus being reached in the LCA research community on the preferred approach (Stewart *et al.* 2005; Berger *et al.* 2011; Yellishetty *et al.* 2011; Klinglmair *et al.* 2014; Schneider *et al.* 2014). In part, this reflects the variety of fundamentals that are addressed by the different metrics. Some take a thermodynamic approach to measure intrinsic properties (e.g. Dewulf *et al.* 2007), but take no account of the role that resource scarcity plays in influencing sustainability concerns. Others use mass-balance based metrics to express the relative scarcity of physical stocks (van Oers *et al.* 2002; Hauschild *et al.* 2005; Schneider *et al.* 2011), but have limited capacity to account for the importance of declining ore quality, nor the possibility that supply-demand changes inevitably encourage the discovery of new deposits. Other recent approaches strive for a compromise, using metrics based on changing ore body quality to incorporate both scarcity and resource quality constraints (Goedkoop *et al.* 2009; Vieira *et al.* 2012; Swart *et al.* 2013).

Because of the different fundamentals that are employed, the relative importance of specific minerals can vary greatly depending on the chosen metric (Goedkoop *et al.* 2009; Swart *et al.* 2013; Klinglmair *et al.* 2014; Schneider *et al.* 2014). It is therefore likely that studies focussed on specific minerals could reach very different conclusions, depending on the metric that is employed.

Of the many LCA models that purport to assess mineral resource sustainability, only three (to our knowledge) provide characterisation factors for mineral phosphorus resources, and are there capable of incorporating biosolids phosphorus recovery into the overall analytical framework. One of those – the EPS2000 metric (Steen 1999) – relies on an assessment of implications in the distance future when concentrated ore bodies no longer exist. This approach appears to have gained little traction in the LCA research community. The other two – CML (van Oers *et al.* 2002) and EDIP (Hauschild *et al.* 2005) – both use simplified metrics based on use : stock ratios that can provide little perspective on the importance of declining ore qualities.

To understand whether phosphorus recovery would appear more or less substantial using some of the fundamentally different metrics available to LCA practitioners, a characterisation factor (53 kg-Fe-e/kg-P) for phosphate rock resources was developed for use with the ReCiPe metric for assessing *Minerals Depletion* (Goedkoop *et al.* 2009), derived from publicly available phosphate rock mining data (Jasinski 2009). The ReCiPe metric is one of a group of contemporary approaches to resource depletion assessment that attempt to incorporate economic principles into the calculation structure.

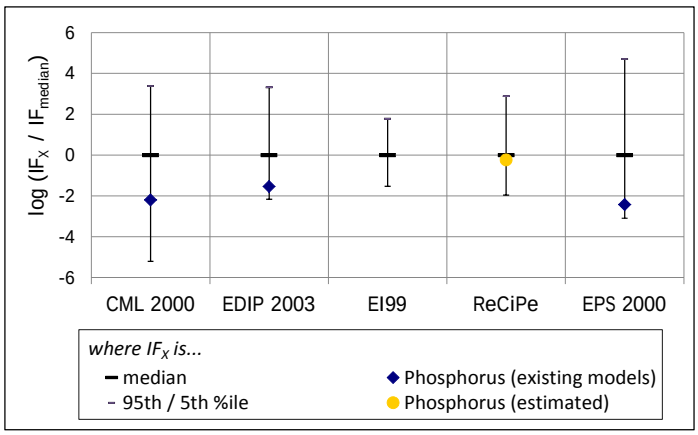

Figure s2: The spread of Minerals Depletion impact factors for different LCIA models, including the three available models that account for mineral phosphate resources - CML (van Oers et al. 2002), EDIP (Hauschild et al. 2005) and EPS (Steen 1999); and the ReCiPe-based model (Goedkoop et al. 2009) amended with our interim factor for phosphate rock. In all cases, phosphorus depletion is considered less significant than the median substance on a unit-mass basis. However, the relative importance of phosphorus under the ReCiPe metric (close to the median) is much greater than under the alternative metrics (~2 orders less significant than the median).

Comparing the spread of midpoint-level impact factors illustrates the significance of choosing different LCA metrics for assessing minerals resource sustainability (Figure s2). Our interim impact factor for phosphate rock is slightly less than the median of all impact factors in the ReCiPe *Minerals Depletion* model, whereas phosphorus resources are considered two orders of magnitude less significant than the median substance under the CML metric (and also under the two other impact assessment models that account for phosphorus resources). In other words, using those alternate *Minerals Depletion* metrics would make the phosphorus recovery seem *even less*, rather than *more*, important.

At face value, the consistently low (less than median) characterisation factors for phosphorus might suggest that phosphorus resource security might not warrant any special priority in broader debates on resource sustainability. However, the comparison presented here gives no indication of whether or not a similar conclusion would be reached if phosphorus characterisation factors were available for some of the quite different metrics that have more recently become available.

Furthermore, consideration should be given to whether LCA resource depletion metrics can adequately represent the somewhat unique socio-economic characteristics of the phosphorus resource sustainability challenge. Because of the fundamental and irreplaceable role of phosphorus in food production, concerns over maintaining equity of access (across poorer countries and/or demographics) feature strongly in debates on this topic (Cordell *et al.* 2009). To our knowledge, there is little, if any, precedent for including such social objectives in metrics developed for LCA resource assessment.

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