Application of Life Cycle Assessment to Wastewater Systems Planning

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

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Description: South Caboolture Sewage Treatment Plant, prior to the plant upgrade implemented in 2012
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FOREWORD

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

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

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

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

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

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

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

Chris Davis
Chair, Urban Water Security Research Alliance

CONTENTS

LIST OF FIGURES

LIST OF TABLES

EXECUTIVE SUMMARY

The Urban Water Security Research Alliance (UWSRA) project *Evaluation Methods for Evidence-based Total Water Cycle Management Planning* investigated a number of modelling approaches that could inform the Total Water Cycle Management Planning (TWCMP) process. As a component of that project, research was undertaken into whether, and how, the Life Cycle Assessment (LCA) methodology could be used to inform the planning for urban water systems infrastructure in South East Queensland (SEQ). This report is one of three produced to meet that goal.

In this particular report, the focus is on conventional wastewater systems and the role that LCA could play in informing key decisions for the wastewater industry. Two case studies are used to highlight the possible benefits, constraints and data requirements in assessing different options for wastewater nutrient removal and biosolids disposal.

A review of recent scientific developments is provided on fugitive greenhouse gas (GHG) emissions from the wastewater system. There has been substantial recent progress in fundamental knowledge about the level of, and drivers for, fugitive emissions from wastewater system infrastructure. While this has greatly improved the industry's capacity to understand the scale of *overall* fugitive emissions from wastewater systems, there remain some important barriers to use of this information for detailed *options analysis*. Research on emissions from receiving environments of wastewater or biosolids is much less advanced, yet the case study analysis illustrates that this could be equally important in the decision making context. With regards to both infrastructure and natural systems, future research should move beyond the characterisation of gross emissions; information is required that can support simplified, but quantitative, assessment of different wastewater management options.

A second methodological review identifies that fugitive emissions, particularly N_2O, are also of great relevance to management of the ozone layer. The link between N_2O and ozone layer depletion is well founded in the scientific literature, and the issue has recently been recognised by the international policy community. Should this translate into requirements for management action, then our analysis suggests that ozone layer depletion might become another environmental concern that water system planners will need to consider in the future.

It would seem prudent for the urban water industry to minimise future risk by avoiding increases in N_2O emissions. However, it cannot be assumed that GHG accounting would be sufficient to meet this objective. Many of the decisions challenging the wastewater sector will involve trade-offs between energy use, N_2O, and emissions of other fugitive greenhouse gases. Hence a decision that reduces the overall GHG footprint for a utility, could still be increasing the overall N_2O flux, and increasing its contribution to the ozone depletion problem.

The investigation into fugitive emissions and ozone layer depletion highlights the strength of LCA, which is the framework that it provides for considering a broad spectrum of environmental issues in a robust and transparent manner. This is demonstrated in a more applied sense through a case study on sewage treatment plant nutrient removal. That case study illustrates how LCA could be used to inform debate on the broader implications associated with delivering high levels of STP nutrient removal. While local water quality concerns have been the dominant driver of strategic environmental thinking in the Australian wastewater industry, this is likely to be questioned as water utilities begin to focus on other environmental considerations (such as GHG emissions).

The LCA framework would be equally relevant to analysis of different biosolids management options, given the diverse environmental considerations involved in the different approaches to biosolids reuse. Unfortunately, some of the biggest shortcomings in the available LCA impact assessment models are also the most relevant for analysis of biosolids reuse in the Australian context. For LCA analysis of soil toxicity, eutrophication and minerals depletion, these shortcomings are likely to bias results against the biosolids reuse option. Substantial improvements to the available models would be required to overcome this, and better match LCA results with the current state of scientific thinking in each of those environmental disciplines.

Proactively addressing the important gaps in the available LCA impact models could help to alleviate some risk for the urban water industry. The Australian agricultural industry is beginning to embrace the use of LCA, and it is possible this will translate into LCA being used to assess different soil and nutrient management options. If that sort of analysis were to (unreasonably) bias against the biosolids reuse option, then there could eventually be a conflict between the needs of the agricultural industry, and the trend towards an increasing reliance on agriculture to absorb urban biosolids.

1. INTRODUCTION

Project Context

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

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

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

The three reports are:

Application of Life Cycle Assessment to wastewater systems planning (this report) provides a detailed investigation into the challenges of using LCA to analyse conventional urban sewage management systems. In particular, LCA is applied to two case studies - analysis of STP nutrient removal and biosolids disposal.

Life Cycle perspectives on wastewater recycling (Lane *et al.* 2012a) provides detailed analysis of wastewater recycling options. While focussed on the Caboolture area, that report aims to inform a broader debate on the role of wastewater recycling in meeting the water supply needs for the rapidly growing population in SEQ.

Using Life Cycle Assessment to inform Total Water Cycle Management Planning – an investigation for the Caboolture catchment (Lane *et al.* 2012c) focuses on the Total Water Cycle Management (TWCM) planning process and directly applies LCA to a specific TWCMP case study for the Caboolture region in SEQ, in order to illustrate the broader benefits and challenges in adopting such an approach. It also provides a synthesis of the findings across all three reports.

Wastewater Systems and the Caboolture TWCMP Study

The Moreton Bay Regional Council (MBRC) has recently undertaken a Total Water Cycle Management Planning (TWCMP) exercise for the urban area within its jurisdiction (WBM 2012, Lane *et al.* 2012c). The TWCMP process aims to encourage strategic decision making that considers all aspects of the urban water system, so as to optimise the urban water system planning process. The Queensland Environmental Protection Act requires that all SEQ councils undertake such a planning process.

At the same time that the TWCMP was underway, an upgrade was implemented at the South Caboolture sewage treatment plant (STP), greatly increasing the level of nutrient removal that it can achieve. As one of the defining goals for the MBRC planning study was to minimise any future increase in nutrient discharges to the Caboolture River estuary, the TWCMP assumed that STP effluent nutrients from the plant would be maintained at low levels over the study lifetime.

By factoring the STP effluent discharge into the planning for regional goals, the TWCMP appears to be consistent with the regulatory requirement that local government TWCM Plans include provisions about "the collection, treatment, and recycling of wastewater". However, by taking the nutrient removal upgrade as a given, the conventional wastewater system (sewerage, sewage treatment and biosolids disposal) was effectively excluded from the TWCM options analysis for the Caboolture area. As a result, the TWCMP study was unable to explore any broader implications of this policy direction.

Also excluded from the TWCMP analysis was any consideration of changes to biosolids management at the South Caboolture STP. As recommended in Unity Water's Biosolids Management Strategy, agricultural reuse is now the preferred disposal method, replacing the previous practice of using biosolids as a bulk fill material. Once again, the scope of the TWCMP study undertaken for the Caboolture area meant that it could not consider the implications of this important change to biosolids management.

The Need for Further Analysis

Unlike the approach taken for the Caboolture area, TWCMP studies in other regions may well need to consider alternatives for wastewater systems management as a part of the options analysis. Our Caboolture-based case study demonstration (Lane *et al.* 2012c) will not provide sufficient guidance on the implications of using LCA to support such a planning study.

Furthermore, it seems likely that choices on data and methodology, relating specifically to wastewater systems, would be critical to analysis for the overall water system. Previous studies have highlighted that the wastewater sector can make a significant contribution to the overall environmental burden of urban water systems, if analysed using the LCA methodology (Lundie *et al.* 2004; Lassaux *et al.* 2007; Lane *et al.* 2011; Lane *et al.* 2012c).

Additional information is required to illustrate the implications and challenges in using LCA to support TWCMP that includes different options for wastewater systems management. The following objectives were defined for this report to help address that gap:

1. *Identify key information that LCA can generate to support environmental analysis of wastewater treatment systems.*
2. *Identify key data requirements for LCA to be implemented, and any gaps in the available data and methodology that would constrain such analysis.*
3. *Identify best-available information/assumptions that can be used for quantitative, broad spectrum environmental analysis undertaken by the wastewater sector.*

Key Gaps in the Analytical Framework

Previous UWSRA research (Lane *et al.* 2011) already identified a number of important challenges for practitioners wishing to apply LCA to wastewater systems analysis. To further its contribution to Objective 3, this report also provides an updated discussion on a number of these concerns.

The most prominent of these challenges relates to fugitive emissions of nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2). Early research in the field identified both the potential significance of these gases to the overall greenhouse gas (GHG) burden of urban wastewater systems, but also highlighted the considerable uncertainties that exist on the topic (de Haas *et al.* 2009; Foley *et al.* 2010a). Since those earlier investigations, the scale and rate of research into fugitive GHG emissions has increased greatly. **Chapter 2** reviews some of these recent scientific developments, focussing on how this science might be applied in a planning context. It also provides detail on the fugitive gas assumptions used for the case study analysis (Chapters 5 and 6), and in the wastewater systems analysis of the two companion reports from this project (Lane *et al.* 2012a; Lane *et al.* 2012c).

Recent years have also introduced a new twist in the context of fugitive gas emissions, with a number of studies highlighting the important effect that N_2O has on depleting the stratospheric ozone layer (Ravishankara *et al.* 2009; Fleming *et al.* 2011; Portmann *et al.* 2012; Revell *et al.* 2012). This suggests that the conventional wisdom, being that the ozone depletion issue has been adequately addressed by global bans on CFCs and other substances, may not be accurate. **Chapter 1** reviews the status of the scientific and policy debates on this topic, in order to highlight its potential relevance to the urban water industry.

For three of the other analytical challenges (Eutrophication, Ecotoxicity and Minerals Depletion) identified by Lane *et al.* (2011), **Chapter 4** provides additional comment on the state of LCA modelling science in each case. These descriptions underpin key parts of the case study discussion presented in Chapters 5 and 6, providing a further contribution to Objective 3 of this report.

Objectives 1 and 2, and to a partial extent Objective 3, are met by applying LCA to analysis of STP nutrient removal (**Chapter 5**) and biosolids disposal (**Chapter 6**). Both case studies are defined for systems of relevance to the Caboolture area of the Moreton Bay Regional Council.

2. FUGITIVE GREENHOUSE GAS EMISSIONS

Lane *et al.* (2011) identified a number of important challenges for practitioners wishing to apply LCA to wastewater systems analysis. One of the more prominent challenges relates to fugitive GHG emissions. Direct (fugitive) emissions of nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2) from the wastewater system are increasingly recognised as a substantial concern for GHG accounting in the urban water industry (de Haas *et al.* 2009; Foley *et al.* 2010a; Lane *et al.* 2011).

Early research revealed the scale of emissions, but also the considerable uncertainties surrounding almost all facets of this topic (e.g. Foley *et al.* 2007; Guisasola *et al.* 2008; de Haas *et al.* 2009; Foley *et al.* 2009; Foley *et al.* 2010a; Foley *et al.* 2010b). More recently, the scientific knowledge base on wastewater-related fugitive gas emissions has improved substantially, particularly in terms of the mechanisms responsible for the emissions that are being observed.

Table 2-1: List of emission factors.

	Source	Our Assumption		Basis	NGERS
C	non-biogenic sewage carbon	9	% of total sewage C	Law (2012)	excluded
	carbon sequestration				
	- biosolids to landfill	modelled		NGERS (DCCEE 2011)	excluded
	- biosolids to agriculture	0.24	fraction of C applied to soils	Brown *et al* (2010)	
CH_4	sewer rising mains conc'n	5	mg-CH_4/L	various	excluded
	fraction of sewer flow through rising mains	100	% of sewer flow		
	STP				
	- aerobic processes	various		Foley *et al* (2010a)	excluded
	- anaerobic processes	modelled		de Haas *et al* (2009)	various methods
	biosolids				
	- interim stockpiling	excluded			excluded
	- to agriculture	2.8	g-CH_4 per kg-ds applied	Foley *et al* (2007)	
	- to landfill	modelled		NGERS (DCCEE 2011)	included
N_2O	STP secondary treatment	8.5	g-N_2O per kg-ΔN^2	Foley *et al* (2010b); Ahn *et al* (2010)	15.7 g-N_2O per kg-N denitrified
	wastewater				
	- to freshwater	2.4	g-N_2O per kg-N discharged	Foley *et al* (2007)	15.7 g-N_2O per kg-N disch
	- to estuaries	9.4	g-N_2O per kg-N discharged		3.9 g-N_2O per kg-N disch
	- to ocean	0.8	g-N_2O per kg-N discharged		0.0 g-N_2O per kg-N disch
	- irrigation	12.6	g-N_2O per kg-N discharged		excluded
	biosolids				
	- to landfill	6.9	g-N_2O per kg-N applied	de Haas *et al* (2009)	excluded
	- to agriculture	15.7	g-N_2O per kg-N applied	Foley *et al* (2007)	
	fertilisers (avoided)	15.7	g-N_2O per kg-N applied	IPCC (2006a)	excluded[3]

[1] values shown here are taken from the NGERS recommended emission factors, recalculated to use the same units as the emission factors used in this study.
[2] $\Delta N = TKN_{INFLUENT} - TN_{EFFLUENT}$.

The uncertainties associated with wastewater-related fugitive gas emissions have been a barrier to practical use of the available knowledge on estimating fugitive emissions. This chapter provides a brief review of the state of the science in fugitive emissions across the wastewater life cycle, in order to capture the most recent developments. The goal is to (a) provide some insight into the possible outcomes as scientific knowledge progresses; and (b) identify ways in which this might be particularly relevant to the analysis required for wastewater systems planning and options analysis.

This chapter also explains the assumptions used to estimate fugitive gas emissions in the case study analysis presented in subsequent chapters (Table 2-1). This information directly addresses Objective 3 of this report. Similar summaries for fugitive gas emissions from the water supply, and stormwater, components of the urban water system are provided in a companion report (Lane *et al.* 2012c).

2.1. Carbon – General Issues

2.1.1. Non-Biogenic Sewage Carbon

Background

The conventional approach to GHG accounting for the wastewater industry is to assume that all sewage carbon is biogenic in origin, and therefore makes a zero net contribution to global warming (IPCC 2006b). 'Biogenic' implies being a part of the natural carbon cycle, whereby carbon is taken from the atmosphere to fuel plant growth, then returned to the atmosphere when that plant matter decomposes. In the case of wastewater systems, the assumption is that 100% of the carbon is sourced from food production, hence the carbon uptake by plants grown for food production will balance out the CO_2 emissions that occur when the sewage is treated. On this basis, sewage sourced CO_2 emissions have been excluded from GHG accounting protocols.

However three recent studies have identified that a significant portion of carbon in urban wastewater systems can be from non-biogenic (e.g. fossil) origins (Griffith *et al.* 2009; Nara *et al.* 2010; Law 2012). It is thought that non-biogenic carbon might come from a range of chemical products, with the biggest contributions being associated with industrial wastewater discharges into the sewer (Law, 2012). Mineralisation of any such carbon in the wastewater treatment process should be included in life-cycle GHG footprints.

The limited available data suggests that sewage composition, rather than differences in treatment technologies, will be the key determinant of how much non-biogenic sewage carbon eventually reaches the atmosphere. For four different treatment trains in use in SEQ, the non-biogenic and biogenic carbon sources partitioned across the aerobic STP unit operations in a similar manner (Law, 2012). The only substantial difference was observed for the one plant with an anaerobic digester, where a marked difference was observed between the partitioning of biogenic and non-biogenic carbon across the digestion operations. Nonetheless, in all cases, the majority (>90%) of non-biogenic carbon was mineralised to CO_2 or ended up in the waste solids stream.

Modelling Approach

Our calculations of non-biogenic carbon emissions are based on an estimate for non-biogenic carbon in the sewage. We also accounted for the origin (biogenic vs. non-biogenic) of any chemically dosed carbon sources into the STP. The non-biogenic fraction of sewage carbon was set at 9% (Table 2-1), being the average result from the four STPs tested by Law (2012). To our knowledge, no other such data on sewage carbon is available in the literature.

As the majority of STPs considered in this project only use aerobic treatment steps, we assumed (in all cases) that biogenic and non-biogenic carbon would behave in the same way across the STP treatment stages. To implement this, we calculated an overall biogenic:non-biogenic ratio across all the carbon inputs, and applied this ratio uniformly to all carbon outflows (wastewater; biosolids; CO_2; CH_4) from the STP. A modification to this approach might be warranted for detailed analysis of treatment plants that use anaerobic digestion to process waste solids.

2.1.2. Carbon Sequestration

Background

Data collected for a previous audit of STPs in the SEQ region (de Haas *et al.* 2009) indicates that the fraction of sewage carbon leaving in the biosolids of BNR treatment plants can be as high as 70%. The method of biosolids disposal is therefore an important determinant to the ultimate fate of sewage carbon, dictating the long term split between sequestration, mineralisation and methane emissions.

As discussed in the previous section, the biogenic portion of any biosolids carbon that is mineralised (to CO_2) is excluded from GHG accounting protocols. But the long-term incorporation of biogenic-sourced carbon into soils would represent a net removal of CO_2 from the atmosphere, and therefore should be given a credit in life-cycle GHG analysis.

However, fossil (non-biogenic) carbon that enters the wastewater system should be handled in the opposite way. In this case, mineralisation of non-biogenic carbon is included as a net emission of CO_2, whereas long term sequestration is excluded from the GHG estimates.

Landfill Disposal

For biosolids that are incorporated into landfill, or used as bulk fill material in other ways (e.g. for mine site rehabilitation), a substantial portion of the biosolids carbon will remain in the ground over the long term. For this study, carbon sequestration under these conditions was estimated using the NGERS carbon decay model for landfill treatment of organic waste material (DCCEE 2011).

It should be noted, however, that for NGERS reporting requirements, this calculation approach is only used for estimating CH_4 emissions. The remaining portion of the landfill carbon balance is excluded, hence the carbon sequestration issue is not considered under the official framework.

Agricultural Application of Biosolids

Biosolids carbon sequestration in agricultural soils is also not considered under the NGERS, nor IPCC, protocols. Furthermore, there is little guidance available on generalised assumptions for use under Australian conditions.

The direct addition of external organic matter (including biosolids) to agricultural soils is considered one of the best ways to increase soil carbon stocks in Australia (Sanderman *et al.* 2010). Supporting this, a number of recent Australian studies have shown that biosolids application can lead to a marked short term increase in the soil organic carbon content (e.g. Powell *et al.* 2012). However, we are not aware of any Australian studies that have demonstrated this can be maintained over the long term. Nonetheless, it seems reasonable to assume that agricultural biosolids reuse could potentially deliver long term sequestration of carbon.

For this study, we assumed that 24% of the biosolids carbon remained in the soil over the long term. This value is based on the modelling methodology proposed for the Canadian water industry (Brown *et al.* 2010), and is used as a means to test the potential significance of this issue.

However, the relevance of this assumption to Australian conditions is unclear, given that warmer Australian temperatures will lead to higher degradation rates. It has been argued that, for Australian soils, carbon sequestration from the direct addition of organic matter might well be lower than in equivalent overseas studies (Sanderman *et al.* 2010).

For the water industry to claim sequestration benefits with some degree of confidence, there is an important need for longitudinal studies that can identify the long term effects on soil carbon resulting from agricultural biosolids use. Guidance will also be required on the sensitivity of sequestration potential to spatial variation in physical conditions (e.g. soil type, crop type) and variation in agricultural management practices.

Wastewater Irrigation

The potential for long term sequestration of carbon in irrigated wastewaters was not considered in this project. Residual carbon in secondary treated effluent would typically represent only ~5% of the total carbon input to the wastewater system. For this reason, omission of this GHG pathway is not expected to have a significant bearing on GHG footprinting in our analysis.

2.2. Methane

2.2.1. Methane Generation in Sewers

Background

Substantial methane (CH_4) generation has been observed in sewer networks by a number of recent SEQ studies (Guisasola *et al.* 2008; Foley *et al.* 2009; Guisasola *et al.* 2009), and in the raw sewage entering STPs in other Australian and international locations (Wang *et al.* 2011; Daelman *et al.* 2012; Law *et al.* 2012a). This contradicts the IPCC and NGERS accounting protocols, which assume that the potential for CH_4 generation from sewer systems is negligible.

The main cause of sewer generated CH_4 is most commonly thought to be anaerobic bacterial activity occurring in rising mains. A simplified empirical model for predicting rising main CH_4 generation has been developed using field data collected at the Gold Coast (Foley *et al.* 2009). This was shown to calibrate well with a more comprehensive model that characterises biological and physicochemical processes in sewers, hence Foley *et al* (2009) argued that their model could be used for estimating CH_4 generation at the scale of STP sewer networks. However, it should be noted that the validity of their empirical relationships has not been tested at a range of temperatures and other key process conditions.

Foley *et al* (2009) also presented evidence of CH_4 generation in open gravity systems, although this phenomenon is less well understood and has barely been studied in the literature. There are no equivalent models available that could be used for estimating CH_4 generation from such systems.

Broader validation of the CH_4 generation model for rising mains, and further testing of gravity systems, forms the basis of current research underway at the University of Queensland (Liu 2012). It is envisaged that this project will enable a greatly improved determination of network scale sewer CH_4 generation rates.

Case Study Application

In the interim, our previous LCA study for the Gold Coast urban water system assumed that network CH_4 generation would average 5 mg-CH_4 per litre of dry weather flow (Lane *et al.* 2011). This was based on the knowledge generated through extensive data collection for Gold Coast sewers (Guisasola *et al.* 2008; Foley *et al.* 2009; Guisasola *et al.* 2009), and the prevalence of pressurised (rising main) sewers in the Gold Coast sewer network.

However, that approach does not provide a precise estimate for the entire Gold Coast network, since the number of sewer monitoring stations was relatively small. Measured sewer CH_4 concentrations at the Gold Coast have been as high as 7 mg/L, with Foley's model (Foley *et al.* 2009) predicting up to 12 mg/L for rising main configurations that are within the typical range for Australian utilities.

The sewer network in the catchment of the South Caboolture STP also contains many rising mains, and it seems likely that sewer CH_4 generation would be occurring. To explore the potential significance of sewer CH_4 in this case study, we used the same 5 mg-CH_4 per L-ADWF assumption as described above. In the absence of any published data on this issue, the biogenic vs. non-biogenic breakdown of the CH_4 carbon was set at the same overall ratio calculated across all STP inputs (section 2.1.1).

Sewer Methane Fate

Finally, we assume that 100% of the sewer generated CH_4 is actually stripped to atmosphere.

There is very little published data available on the fate of dissolved sewage CH_4 generated in sewers. One Dutch study showed that the activated sludge process steps could oxidise up to 80% of the dissolved CH_4 in the plant feedstock, thereby avoiding significant CH_4 emissions (Daelman *et al.* 2012). In contrast, interim results from testing at one particular Australian STP indicate a much higher fraction of dissolved CH_4 in the raw sewage being stripped within the STP boundary (Law *et al.* 2012a).

Importantly, neither of those studies provides insight on the potential for significant portions of sewer generated CH_4 to be stripped at points along the sewer network, prior to reaching the STP. This seems entirely plausible, given the likely prevalence of intermediate pumping stations and gravity sections in large sewer networks.

For effective sewer CH_4 accounting, further research should therefore characterise the potential for dissolved CH_4 stripping (or oxidation) both within the sewer network and within different STP configurations.

Non-Biogenic CH_4

We assumed that 9% of all sewer methane would be of non-biogenic origin, using the same data described in section 2.1.1.

2.2.2. CH_4 Generation in Sewage Treatment Plants

CH_4 generation in sewage treatment plants is primarily associated with anaerobic treatment processes. For those few MBRC treatment plants that do include anaerobic treatment steps, we used a mass balance based methodology for estimating CH_4 losses that was developed by de Haas *et al.* (2009). For managed aerobic treatment steps, the default assumption in IPCC guidelines and the NGERS protocol is that there will be no CH_4 emissions. The IPCC guidelines do, however, note the possibility that CH_4 generation could be as high as 10% of COD removed from the wastewater.

The South Caboolture STP (our principle focus in this study) does not include any anaerobic treatment steps, hence CH_4 generation within the STP is not expected to be a significant source of GHG emissions. Nonetheless, to account for the possibility that small amounts of CH_4 can be generated from aerobic treatment trains, we have used emission factors for CH_4 generation extrapolated from the STP modelling of Foley *et al.* (2010a).

Section 2.1.1 describes the calculation approach used to estimate the fraction of this carbon that is of non-biogenic origin.

2.2.3. CH_4 from Biosolids Disposal

Landfill Disposal

Under the NGERS protocol, CH_4 emissions from biosolids disposal to landfill can be estimated using a 1st order carbon decay model (DCCEE 2011). An alternative approach would be to use emission factors based on a literature data (e.g. Foley-WSAA or Brown).

The mechanistic NGERS model seems the most appropriate way to factor in spatial variation across different landfill sites, and so was adopted for the analysis in this project.

Another critical assumption used in this project was that none of the landfill-generated CH_4 is captured for power generation. While there are Australian landfills with a high degree of CH_4 recovery, this practice is not that widespread in Australia. The conservative assumption (zero recovery) was used to indicate the potential scale of this flux to GHG accounting for sewage treatment plants.

Agricultural Application

Neither the NGERS, nor IPCC, accounting protocols require consideration of CH_4 emissions generated from biosolids that are applied to agricultural topsoils. To include an estimate for this issue, we used the median value (2.8 g-CH_4 per kg-ds) from the literature review of Foley *et al.* (2007). No further review of more recent literature was undertaken during this project.

However, 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. This issue has been included in the Canadian GHG accounting framework proposed by Brown *et al.* (2010). While this could potentially be an important source of CH_4 emissions, we did not review the prevalence of this practice in Australia.

Non-Biogenic CH_4

Section 2.1.1 describes the calculation approach used to estimate the fraction of biosolids carbon that is non-biogenic. For both disposal pathways considered in this project, this fraction was applied uniformly to all long term carbon pathways (mineralisation; methane release; soil accumulation).

2.3. Nitrous Oxide

2.3.1. Nitrous Oxide from Sewage Treatment Plants

Background

The conventional water utility approach to estimate nitrous oxide (N_2O) generation from STPs is to follow the NGERS GHG accounting framework (DCCEE 2011). However the NGERS emission factor (15.7 g-N_2O/kg-N denitrified)[1] was developed at a time when there where was very little N_2O emissions data available.

Recent years have seen a significant increase in research effort to quantify N_2O emissions from STPs, and to understand the fundamental mechanisms leading to N_2O creation. However, there remains a significant degree of uncertainty about how to best estimate these flows (GWRC 2011; Law *et al.* 2012b). What is well recognised is the considerable variation that exists across different treatment plants, with published N_2O generation rates varying from 0 to 25% of total influent nitrogen (Law *et al.* 2012b). Within a single plant, substantial variations over time (greater than tenfold) have also been observed as a result of diurnal changes in process conditions and/or process perturbations (Ahn *et al.* 2010; Foley *et al.* 2010b). A recent review across four international studies concluded that this large variation (over time, and across STPs) can likely be explained by differences in: (a) the extent that nitrogen concentrations vary along the treatment train; (b) variability in the influent loads presenting to the STP reactor; and (c) maximum nitrite concentrations encountered across the treatment process (GWRC 2011).

It is thought that the more stable operating conditions (often) found in STPs with larger throughput, and/or greater levels of nutrient removal, might serve to reduce the overall conversion rates of sewage nitrogen to N_2O (Foley *et al.* 2010b; GWRC 2011; Law *et al.* 2012b). It should be noted, however, that the evidence supporting this conclusion is based on a comparison across *different treatment plants*. There is no such evidence showing that *upgrading an existing STP* (to process higher flows and/or deliver a higher degree of nutrient removal) would *necessarily* result in a lower proportion of the nitrogen removal ending up as N_2O. The extent, to which an STP upgrade would change the N_2O emissions intensity, would depend on the effect it has on the key process conditions identified above.

N_2O Emission Estimates

The most commonly used literature approach to benchmarking N_2O emission rates is to express the N_2O generation as a function of the nitrogen (TKN) load in the sewage (Ahn *et al.* 2010; GWRC 2011; Desloover *et al.* 2012; Law *et al.* 2012b). Given the uncertainties involved, this simplified approach might be adequate for generating overall emissions estimates across the industry as a whole.

However, the science clearly indicates that N_2O generation is a function of process conditions, rather than being a function of sewage load. Emission factors based on sewage nitrogen loads would therefore not support comparison of systems involving fundamental differences in the nitrogen balance within the wastewater system. For example, they could not be used to explore the implications of different effluent nitrogen concentrations, different sludge generation rates, and/or different effluent and sludge disposal pathways. This type of analysis requires a more mechanistic approach to estimating emission levels, in a manner that can reflect differences in an STP nitrogen mass balance.

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[1] reported as 4.9kg CO_2-e/kg N-denitrified, derived from 0.0157 kg-N_2O/kg N-denitrified × 310 kg CO_2-e/kg N_2O

Emission factors reported (Foley *et al.* 2010b) and used (de Haas *et al.* 2009) in recent Australian studies were scaled to the amount of nitrogen that was denitrified (emitted as gas) in the treatment plant. This approach is consistent with the calculation methodology adopted by NGERS (DCCEE 2011), and reflects the historical perception that the denitrification pathway was the main source of N_2O emissions from STPs.

However, N_2O can be generated in both the nitrification and denitrification steps, and recent investigations have shown that process conditions during the nitrification stage are the most important determinant of overall N_2O emission rates (Law *et al.* 2012b). This suggests that a denitrification based emission factor may not correlate well with the key causative factors of N_2O generation in many STPs.

Case Study Application

For the analysis in this report, we have used an emission factor scaled to the mass of nitrogen removed from the wastewater (*Equation 2-1*), as the simplest alternative proxy for the level of nitrification/denitrification activity occurring in the STP. This value (8.5 g-N_2O/kg-N removed from wastewater) was set at the median of estimates for nineteen STPs, taken from the two largest studies available in the literature. One of these studies measured N_2O generation from twelve STPs in the USA, using a technique for continuous monitoring of gas fluxes from the treatment plant reactors (Ahn *et al.* 2010). The other study estimated N_2O generation from seven Australian STPs using grab sample measurements (Foley *et al.* 2010b). Emission factors (per mass of nitrogen removed from the wastewater) were calculated from the raw data collected for that study.

$$m_{removed}^{N} = m_{sewage}^{TKN} - m_{effluent}^{TN} \qquad \textit{Equation 2-1}$$

It is recognised that this approach will not support robust N_2O analysis of wastewater process options, since it cannot account for the effects of internal recycle streams, nor variation in the process factors identified above. Such analysis would require a more detailed process model, which is the subject of current research efforts at the University of Queensland.

Until process-specific emission factors become available, alternative approaches are required. The challenge is for utilities to ascertain their potential future GHG exposure once more realistic N_2O emission estimates become possible. The risk matrix proposed by GRWC (2011) offers a qualitative means for addressing this question (Table 2-2).

Table 2-2: Key factors for assessing risk of N2O emissions from sewage treatment processes (from GWRC 2011).

Parameter	High Risk	Medium Risk	Low Risk
Effluent total nitrogen (mg/L)	> 10	5 - 10	< 5
Range in N-concentration in plant	H	M	L
Load variations (daily)	H	M	L
Maximum NO_2 concentration (mg-N/L) anywhere in plant	> 0.5	0.2 – 0.5	0.2

While this qualitative guide is valuable, it is not sufficient for planning purposes. Water utilities will require quantitative estimates that reflect the latest science, if they are to benchmark the scale of STP-sourced N_2O emissions against other sources of GHG exposure. Unfortunately, there is insufficient data available to meaningfully apply different emission factors across different treatment processes. We therefore applied a single emission factor (described in *Equation 2-1*) to all STP analysis undertaken in this project.

For the STPs within the MBRC region, use of our emission factor would give N_2O emission estimates 0-30% lower than if using the default NGERS approach. However, it is important to recognise that there remains large uncertainty associated with the absolute value of the emission factor used here.

The 95th percentile of the data surveyed for this report was 68 g-N_2O/kg-N removed, eight times the median value that we used. The significance of this uncertainty will be increased if applying that median value, or any other uniform factor, to comparisons across different STP nitrogen removal technologies, different locations, and/or different degrees of nutrient removal.

2.3.2. N_2O from Biosolids Disposal

For advanced BNR sewage treatment, 20-40% of the sewage nitrogen might typically end up in the solids waste stream. Biosolids disposal therefore represents the second largest flux of nitrogen leaving the sewage treatment system, and could potentially be another substantial pathway for N_2O generation.

Landfill Disposal

Neither the NGERS protocol nor the IPCC provide a calculation methodology for estimating biosolids disposal to landfill. Literature studies have used a wide range of values for biosolids-related GHG accounting, with one study (Brown *et al.* 2010) recommending a value as high as 24 g-N_2O per kg-N applied to the soil. This latter value is not consistent with the more conventional expectation, that N_2O emission rates from well-managed, anaerobic landfill processes would be extremely low (Foley *et al.* 2007).

The reasons for the large variation in the literature are not clear, although may be related to differences in landfill management practices. Further investigation would therefore be required to ascertain the best approach to generating realistic estimates from Australian landfills.

For this study, we used an emission factor (6.9 g-N_2O per kg-N in the biosolids) at the lower end of the spectrum, adopting the median of the literature values for landfill disposal collated by Foley *et al.* (2007).

Agricultural Application of Biosolids

For biosolids disposal to agricultural land, we used an emission factor (15.7 g-N_2O per kg-N applied to soils) recommended by Foley *et al.* (2010a). This matches the conventional approach recommended by the IPCC (2006b), although this pathway is not required for GHG accounting under the NGERS protocol.

However 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). We are not aware of any Australian studies measuring 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 *et al.* 2006). 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 N_2O emissions. An overall emissions balance for agricultural reuse would therefore need to reflect the net difference between N_2O generation caused by the biosolids, and N_2O generation caused by the use of alternative fertilisers. To calculate the latter, we used the conventional approach to GHG accounting for synthetic fertiliser use, taken from the IPCC guidelines (2006a), which is to assume that 1% of applied N (or 15.7 g-N_2O per kg-N applied) is lost as N_2O.

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. The available Australian data shows that agricultural N_2O emissions can vary from substantially lower than the default IPCC value, to as high as 27% of applied N. The potential for soil-generated N_2O emissions is generally considered to increase in warm or humid climates, and where crop irrigation is practiced (Thorburn *et al.* in press).

The analysis for this report used equal emission factors (15.7 g-N_2O per kg-N applied to soils) for both biosolids nitrogen and synthetic fertiliser, consistent with the conventional approach adopted in many other studies. However, it is not clear whether this is likely to be representative of actual outcomes, given that the form of the nitrogen in the two products is fundamentally different. Biosolids nitrogen is predominantly organic, whereas conventional fertiliser is normally applied as ammonium compounds.

Future Analysis

Given the potential variability in N_2O generation rates from agricultural systems, it is clear that GHG analysis of biosolids reuse scenarios should be undertaken with case-specific data as much as possible. Furthermore, meaningful N_2O accounting for biosolids reuse will not be possible without further research into the likelihood of N_2O generation from biosolids amended fields under a range of conditions.

2.3.3. N_2O from Wastewater Disposal

Modelling Approach

The default values for wastewater discharge to waterways, and wastewater irrigation, are based on a previous literature review (Foley *et al.* 2007). The NGERS protocol recommends the use of different emission factors, which are outlined in Table 2-1. No further review has been conducted through this project.

Analytical Implications

While such estimates will be extremely uncertain because of a lack of empirical data, the total contributions to the N_2O profile of a wastewater system are typically very small (Lane *et al.* 2011). However, the relative size of the different emission factors might be an important point to consider.

The proposed emission factors for wastewater discharge to an estuarine environment (9.4 g-N_2O per kg-N discharged) and via irrigation (12.6 g-N_2O per kg-N applied) are very similar, and are also of the same order of magnitude as the two possible emission factors for in-plant nitrogen removal (8.5 g-N_2O per kg-N removed; or 15.7 g-N_2O per kg-N denitrified) for this study. This indicates that, if using these emission factors, the calculated net overall flux of N_2O originating from sewage nitrogen would change very little if analysing a reduction in estuarine discharge of effluent nitrogen, regardless of whether that is achieved by wastewater diversion to irrigation, or by increased STP denitrification.

A very different outcome might eventuate if the STP effluent is being discharged to freshwater (2.4 g-N_2O per kg-N discharged) systems or the ocean (0.8 g-N_2O per kg-N discharged). These factors are much lower than those for other pathways, suggesting that diversion of effluent nitrogen from these disposal points would lead to substantial net increases in the total calculated N_2O flux.

The above discussion highlights that, in the context of planning future changes to wastewater systems, realistic emission factors for effluent receiving environments could be just as important to life-cycle N_2O analysis as are estimates of N_2O fluxes from the STP treatment train. A number of research projects are underway in SEQ, collecting the first local data on N_2O emissions from fresh and coastal waterways (Grinham 2012). These studies may underpin the future determination of locally specific estimates for N_2O generation that result from net changes in wastewater nitrogen discharge.

Irrigation Offsets

While not included in this study, systems that involve large scale wastewater irrigation of agricultural lands should also account for the implications of any displaced fertiliser use. An overview of the issues relating to this aspect is provided in section 2.3.2.

3. NITROUS OXIDE AND THE STRATOSPHERIC OZONE LAYER

This chapter provides a summary of the state of ozone-depletion science, and an indication of the potential relevance of this issue to wastewater systems management. It draws on the discussion presented in an earlier paper published by the project team (Lane *et al.* 2012b). However it also provides an update of subsequent publications in the atmospheric science literature, and extends the analysis to consider the Moreton Bay Regional Council (MBRC) urban water system.

The information provided in this chapter directly addresses Objective 3 of this report. The recommended impact factors for ozone depletion assessment are also adopted in the case study analysis provided in subsequent chapters.

3.1. N_2O Impact on 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). A review of the relevant literature highlights that the reaction pathways, and the significant role of N_2O in depleting ozone concentrations at certain altitudes, are well understood and accepted in the atmospheric science community.

More recent atmospheric modelling has shown that, with the majority of CFC and HCFC emission sources already controlled, N_2O emissions now represent the biggest source of human-caused ozone layer depletion (Figure 3-1).

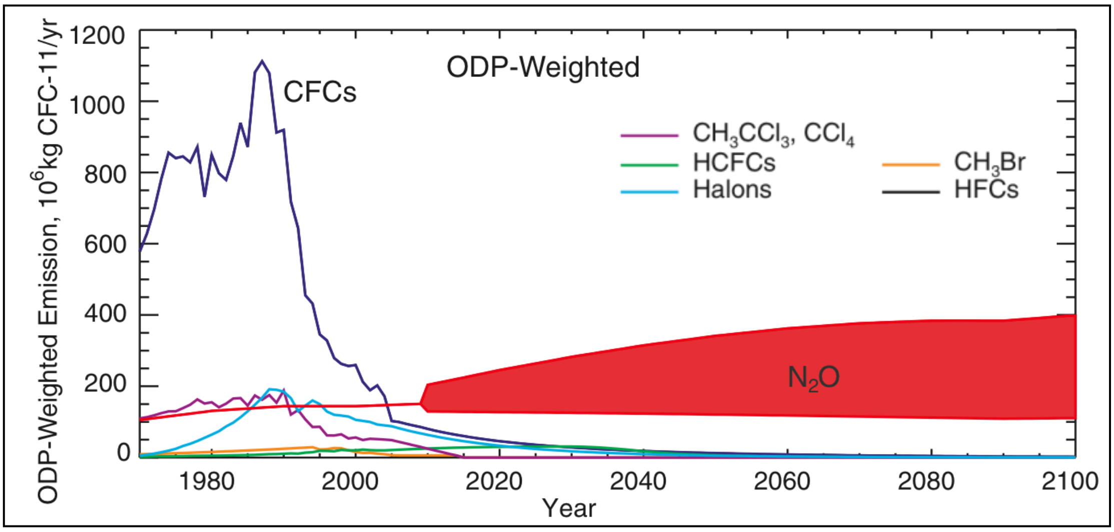

Figure 3-1: Historical and projected ODP-weighted emissions, taken from Ravishankara et al. (2009), using a range of global anthropogenic N2O emission forecasts that were inferred from IPCC scenarios. Even assuming no growth in anthropogenic N2O generation (the lower bound), N2O will be the dominant source of stratospheric ozone layer depletion into the future.

These forecasts were produced by Ravishankara *et al.* (2009) using the first ever estimation of an Ozone-Depletion Potential (ODP) for N_2O (0.017 kg-CFC11 / kg-N_2O). ODP factors are the most commonly used policy metric for assessing the implications of discrete choices that affect ozone layer integrity, are regularly updated by the World Meterological Organisation (WMO) Scientific Assessment studies, and are widely used as the metric by which ozone layer depletion is incorporated into LCA studies. More recent modelling studies, using similar atmospheric conditions, have produced steady-state ODPs for N_2O of 0.019 (Daniel *et al.* 2010; Fleming *et al.* 2011) that are consistent with the Ravishankara *et al.* (2009) proposition. While relatively low compared to many other ozone-depleting substances, the proposed ODP for N_2O is within the range of ODP factors (0.01-0.12 kg-CFC11/kg) assigned to the HCFCs that are currently being phased out under international regulations.

The Montreal Protocol (1987), the chief international policy instrument that has driven legislative change to avoid the use of chlorinated and brominated ozone depleting substances, has never included N_2O in the list of controlled substances. Considering the urgency of the ozone problem that faced the international community in the 1980s, it seems that the long term nature of the ozone effects caused by N_2O were less of a priority than the short term effects caused by chlorine and bromine emissions (Chipperfield 2009; Portmann *et al.* 2012).

However, the most recent WMO Scientific Assessment (WMO 2011) has for the first time included comment on the ozone depleting contribution from N_2O emissions. Furthermore, a recent study has confirmed that N_2O mitigation represents the single largest opportunity for avoiding further depletion of the ozone layer (Daniel *et al.* 2010).

It might therefore be expected that increased policy attention could be given to N_2O management into the future. The nature of any such policy response may well depend on how significant the remaining ozone layer 'problem' is perceived to be. It has been shown that the possible benefits from future N_2O mitigation are small compared to those associated with the reductions in chlorinated and brominated halocarbon emissions already delivered by the Montreal Protocol (Daniel *et al.* 2010).

With the long term ozone implications of N_2O now being included in atmospheric modelling studies, it has become apparent that the positive effect on ozone concentrations caused by climate change is the main reason why global ozone levels are still expected to return to 'normal' levels by the end of this century (Eyring *et al.* 2010; Plummer *et al.* 2010). Given this complexity, it is not yet clear what the WMO might recommend in terms of policy response to the N_2O – ozone depletion challenge.

3.2. Ozone Depletion Modelling for LCA

A recent European Union report recommended that ODP models for LCA be based on ODP factors published by the WMO (ECJRC 2011). However most publicly available LCA impact assessment methodologies still rely on outdated values provided in the 2002 WMO Scientific Assessment (Montzka *et al.* 2003). The ozone depletion models used in this project incorporate the updates provided in the 2006 (Daniel *et al.* 2007) and 2010 (Daniel *et al.* 2011) WMO Scientific Assessment reports.

With the publishing of ODP impact factors for N_2O, it is now possible to include N_2O in LCA ozone depletion analysis. However, in doing so, there are a number of caveats that must be recognised.

Firstly, the available values from the literature were not developed with the same models employed for generating the recommended WMO values, and by inference, the other ODP characterisation factors commonly adopted in LCA. However the model used by Ravishankara *et al.* (2009) was shown to calibrate well with the official ODP values for CFC-12 and HCFC-22. Furthermore, all the available literature values were generating using steady state modelling in a manner broadly consistent with the favoured approach in LCA. It is therefore suggested that any potential differences due to source model variation are likely to be minor.

Far more significant is the fact that 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 significantly influenced by the ambient atmospheric chlorine concentration chosen for model simulations. The available literature ODP values were all calculated for atmospheric conditions in the year 2000, at which time the stratospheric concentration of reactive chlorine was very near its historical peak (Clerbaux *et al.* 2007; Austin *et al.* 2010). Future stratospheric chlorine concentrations will be much lower, and the modelling of Ravishankara *et al.* (2009) shows that this could increase the calculated ODP for N_2O by up to 50%.

Whereas the downward trend in atmospheric chlorine will increase the level of ozone damage done by future N_2O emissions, climate change will have the opposite effect. The conversion of N_2O to reactive nitrogen oxides (that cause the ozone layer damage) is known to decrease with increases in stratospheric temperature. Plummer *et al.* (2010) show that moderate growth in GHG emissions to 2100 would roughly halve the ozone destructiveness of N_2O.

Further guidance from the atmospheric modelling and LCA research communities will be required to determine the most appropriate range of ODP factors for N_2O, if their use is to be consistent with the modelling paradigms underpinning other LCA impact categories. In the interim, LCA analysis should adopt the ODP factor (0.018 kg-CFC11e/kg-N_2O) for N_2O that is used in this study. This was taken as the average of the three values available from the literature.

3.3. Sources of N_2O from the Urban Water System

To ascertain the points of greatest exposure to any policy focus on N_2O, the breakdown of N_2O sources across the urban water system have been calculated for the 2010 MBRC urban water system (Figure 3-2). This model was developed and described in a complementary report produced by the project team (Lane *et al.* 2012c). It provides a snapshot of the overall environmental burden associated with water supply, wastewater and stormwater systems within the MBRC area. N_2O emission factors for the wastewater system are described in Chapter 2.3, and in Lane *et al.* (2012c) for the water supply and stormwater components. "Power generation" in the model refers to emissions associated with the generation of the power used by the urban water system built assets.

Estimation of direct N_2O emissions from urban water systems is an extremely inexact science. However the assumptions used to calculate the N_2O emissions profile across the MBRC system were based on detailed reviews of the available literature, and represent a science-based approach to fugitive gas accounting. Conventional GHG accounting protocols, such as those required for NGERS (DCCEE 2011) or IPCC (2006b) compatibility were not used for this analysis.

The approaches used to estimate N_2O emissions across the wastewater sector are described earlier in this report (Chapter 2). The equivalent assumptions for the water supply and stormwater sectors are described elsewhere (Lane *et al.* 2012c), although two specific points are worth noting.

Firstly, estimates of N_2O emission from dams were excluded despite evidence that this occurs in SEQ. There remains considerable uncertainty on how to characterise the anthropogenic N_2O contribution, and the available data suggests that the anthropogenic N_2O emissions would make only a minor change to this analysis.

Secondly, the potential emissions from existing stormwater management systems were also excluded, through a lack of data on the extent of system use across the region.

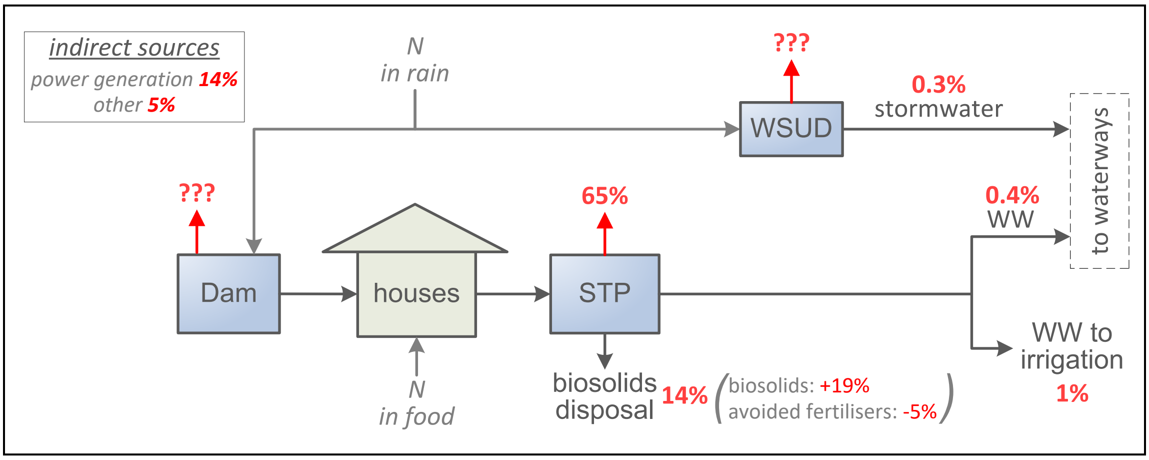

Figure 3-2: Breakdown of N_2O emissions from the MBRC urban water system, which is dominated by STP nutrient removal (76%) and biosolids disposal (8% net).

3.3.1. Direct Sources

Despite the significant uncertainties involved in characterising direct N_2O emissions across the urban water system, it is likely that STP operations provide the major source of direct N_2O in the urban water system. N_2O generated in the nitrification-denitrification processes within sewage treatment systems constitutes 76% of our total estimate (Figure 3-2).

The only other notable contribution (14%) is associated with biosolids disposal (Figure 3-2). Unlike in some other regions of SEQ, the majority of biosolids from STPs in our MBRC model were not used on agricultural fields, and therefore do not displace the use of alternative fertiliser sources. A shift towards substantial agricultural biosolids use (Chapter 6) would greatly reduce the biosolids contribution, as the calculation of net N_2O fluxes would then incorporate credits for the N_2O that would otherwise have been associated with synthetic fertiliser use.

Collectively, these results confirm that the wastewater sector is likely to be the dominant source of direct N_2O across the urban water system. While shown here as a gap, it is not expected that anthropogenic N_2O generated by water supply dams will be notable (Lane *et al.* 2012c), although this remains to be confirmed. The uncertainty associated with the exclusion of stormwater treatment (e.g. Water Sensitive Urban Design) is also unclear, although we expect that the total contribution from such systems would also be low.

3.3.2. Indirect Sources

While only representing 7% of the total N_2O emissions load (Figure 3-2), the N_2O contribution estimated for power generation warrants further consideration. The quality of the database information available for Australian power generation sources needs some review, because there are two situations envisaged where this indirect N_2O contribution could appear substantial. The first relates to the considerable growth in water supply energy intensity that is expected to occur as the population in South East Queensland grows (Hall *et al.* 2011; Lane *et al.* 2011).

The second recognises the constrained system boundary for this study, which excludes the power use for domestic water heating and other water use applications. Other studies have found that the use-related energy component can be in the order of six times greater than that for operating the built infrastructure of urban water systems (Kenway *et al.* 2011). Given the strong influence that water utilities can have on the level of total community water use, it may in fact be that management of indirect energy use (e.g. domestic water heating) could offer the opportunity to mitigate N_2O emissions to a substantial level.

3.4. Significance of Ozone Depletion for Water Systems Planning

In deciding how to respond to the potential concern over ozone layer damage, water planners will require some broader perspective on the relative importance of the N_2O emissions produced by their facilities. To assist with this, the full set of LCA results for the model of the 2010 MBRC urban water system are benchmarked against the equivalent LCA results for the Australian economy (Figure 3-3). Lane *et al.* (2012c) describe both the datasets used for that benchmarking exercise.

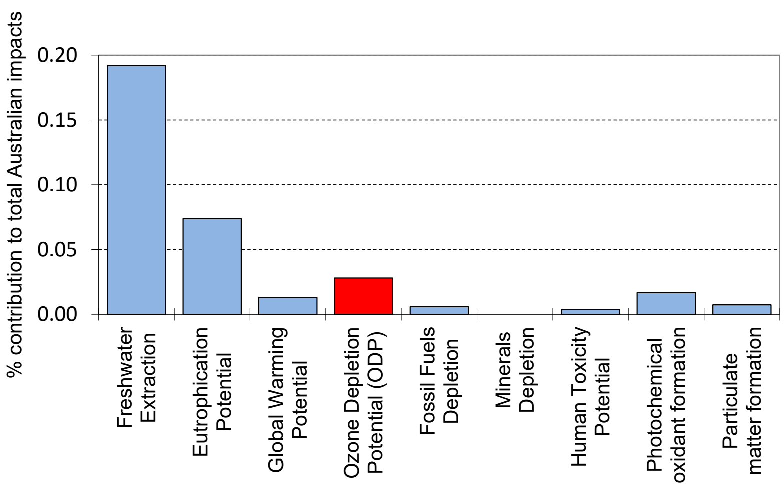

Figure 3-3: Selected LCIA results for the MBRC urban water system, normalised against an interim estimate for overall impact of the Australian economy (Lane *et al.* 2012c).

Figure 3-3 shows the urban water system's contribution to ozone layer depletion to be on a par with issues, such as global warming, that are already of some concern to urban water system planners. The implication is that this might be one of the more significant environmental issues associated with the urban water system, when considered from the broader perspective of Australian society. Care must be taken in drawing conclusions on results normalised in this way, as there is potential for hidden bias to affect the interpretation (Heijungs *et al.* 2007). Nonetheless, if the international community were to focus attention on mitigation of N_2O emissions for the sake of ozone layer protection, the normalised results indicate that Australian policy makers might consider this a relevant concern for the managers of urban water systems.

If this were to happen, then the inclusion of ozone depletion considerations could have a tangible impact on decision making processes under certain circumstances. This is illustrated with a case study that investigates the tradeoffs associated with choosing to improve the nutrient removal performance (from effluent TN=20 mg/L to effluent TN=10 mg/L) for a single STP. Both scenarios employed primary treatment, an MLE activated sludge configuration, and anaerobic digestion of biosolids. In both cases, estimates for inputs (power and chemicals use) and outputs (biosolids, fugitive gas emissions) were taken from Foley *et al.* (2010a).

Arguably, the only environmental 'externality' that might be considered in such a decision making process would be the implications for GHG budgets. For this case study, the 50% reduction in effluent nitrogen would come at the expense of a 19% increase in life-cycle N_2O generation. However, there is only a minor increase in *Global Warming Potential* (Figure 3-4). This is because the overall *GWP* of the sewage treatment plant is heavily influenced by power use, which changes little between the two chosen scenarios.

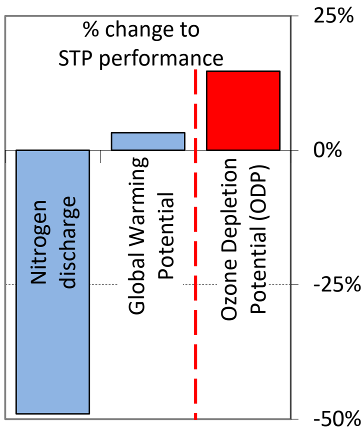

Figure 3-4: Impacts of a hypothetical STP case study considering a shift from effluent TN=20mg/L to effluent TN =10mg/L (both scenarios taken from Foley *et al.* 2010a).

The increased nutrient removal causes only a minor increase in GHG emissions, because the increased N_2O emissions are offset by reduced import of power. However the inclusion of an ODP metric highlights a substantial downside to the enhanced nutrient removal.

If the environmental considerations ended here, then the results would suggest only a negligible trade-off, and therefore great benefits in choosing the STP technology with lower effluent nitrogen levels. However, if ozone layer depletion is included in the comparison, then a potentially significant downside to the nutrient removal is identified. This is, of course, a somewhat hypothetical case study that does not reflect the complexity of real-life decision making that would consider many other issues (economic, social and environmental). However, this example does illustrate that ozone depletion results could affect urban water planning outcomes when there are substantial N_2O emissions involved in the infrastructure under consideration. While these effluent TN concentrations are high by the standards of most SEQ treatment plants, they would be encountered in many Australian locations. More importantly, the case study focussed on the implications of increased nutrient removal, which is a driver for urban STP management across the majority of big and small urban centres in Australia.

3.5. Conclusions – Implications for the Urban Water Industry

While it is not yet clear what form the policy response will take, ozone layer depletion might become a management concern for water utilities should the ozone threat from N_2O become a focus of international policy action. This increases the imperative for N_2O to be actively addressed in urban water systems.

It would therefore seem prudent for the urban water industry to avoid (where possible) strategies that increase the overall intensity of their N_2O emissions. Ideally, changes that can reduce the likelihood of N_2O emissions should be incorporated into water utility strategic plans.

The two areas most likely to yield mitigation opportunities are the management of STP nutrient removal processes, and the management of STP biosolids disposal. The adoption of wastewater treatment technologies, or operating strategies, that provide more stable control of total nitrogen and nitrite concentrations is considered likely to reduce the risk of N_2O generation in the nitrification-denitrification steps of BNR plants (see Chapter 2.3). With regards to biosolids, the most likely avenue for mitigation is to ensure that biosolids disposal can effectively displace the need for alternative fertilisers, while ensuring that excess mineralisation rates of the biosolids organic N are avoided (see Chapter 2.3).

When considering how to respond to this issue, it should be recognised that the inclusion of GHG accounting in decision making processes will not necessarily ensure that N_2O emissions are minimised. Under some circumstances, a focus on overall GHG burdens could mask the situation whereby a reduction in energy use comes at the expense of increased N_2O emissions. The example used in this analysis (Figure 3-4) is just one of the many possible situations where such an outcome could occur in an urban water systems planning context. One simple way to avoid this problem would be to include the *Ozone Depletion Potential* metric (as proposed in this report) in multi-criteria decision making processes. This would guarantee that differences in N_2O emissions are detected in comparative scenario analysis.

Finally, despite a substantial increase in recent research effort, there remain considerable knowledge gaps on the quantity and fundamental mechanisms driving N_2O emissions from urban water system components. Addressing these gaps will be necessary, if the water sector is to develop a robust understanding of the optimal ways to reduce direct N_2O emission levels.

4. LCA IMPACT ASSESSMENT FOR THE CASE STUDY ANALYSIS

The LCA impact models used in this research project were developed through a detailed review of: (a) key limitations with the default methods currently in use; (b) the implications of these limitations for urban water systems analysis; and (c) a survey of current methodological research underway internationally. The majority of these were taken from, or modified from, the ReCiPe impact assessment methodology (Goedkoop *et al.* 2009). Significant changes were made to some of the default LCA impact models during the course of this research project, while for others a number of important challenges were identified.

This report provides details on the reviews undertaken for the modelling of the LCA impact categories *Eutrophication Potential*, *Ozone Layer Depletion* and *Minerals Depletion*. Detailed discussion on *Terrestrial Ecotoxicity Potential* is intended to complement the additional summary of LCA toxicity modelling provided in the companion report focussed on wastewater recycling (Lane *et al.* 2012a).

Table 4-1 summarises those particular impact categories and impact models that were used for the analysis presented in this report, and provides a link to the detailed reviews undertaken for other impact categories. The full suite of LCA impact categories that were considered in this research project are summarised in a companion report to this study (Lane *et al.* 2012c).

Table 4-1: Summary of LCA impact assessment models used in this report for the impact categories.

Impact Category	As proxy for…	Modelling Approach for this Analysis		Detailed Description
Freshwater Extraction (FWE)	ecosystem damage from disruptions to the hydrological cycle, and from discharge of nutrients/COD to waterways	excluded	none of the scenarios have a direct influence on freshwater extraction	*[1]*
Eutrophication Potential (EP)		customised model	account for local waterway sensitivities	*Chapter 4.1* *Chapter 6.3*
Ecotoxicity Potential (marine – METP; freshwater – FETP)	ecosystem damage from discharge of organic and metal pollutants	excluded	model fidelity considered inadequate	*[2]* *Chapter 4.2* *Chapter 6.3*
Ecotoxicity Potential (terrestrial – TETP)		excluded (except in Chapter 6)		
Global Warming Potential (GWP)	ecosystem and human health damage from changes to atmospheric concentrations of greenhouse gases and stratospheric ozone	ReCiPe model		*[1]*
Ozone Depletion Potential (ODP)		modified ReCiPe model	updated ODP factors for conventional ozone depleting substances and an ODP factor for N_2O	*Chapter 3.2*
Fossil Fuels Depletion (FD)	availability of critical resources for use by future generations	ReCiPe model		*[1]*
Minerals Depletion (MD)		modified ReCiPe model	incorporate an interim impact factor for phosphate rock resources (ch.7 only)	*Chapter 4.3* *Chapter 6.3*
Human Toxicity Potential (HTP)	human health damage by emissions of organic chemicals and metals (HTP); and substances that cause smog (POFP), and atmospheric particulate accumulation (PMFP)	modified USES-LCA v2	impact factors for metals were downgraded	*[2]*
Photochemical Oxidant Formation Potential (POFP)		ReCiPe model		*[1]*
Particulate Matter Formation Potential (PMFP)		ReCiPe model		*[1]*

1. see Lane et al (2012c), 2. see Lane et al (2012a)

4.1. Eutrophication Potential Modelling

The *Eutrophication Potential (EP)* metric has long been included in the LCA domain, acting as a proxy for aquatic ecosystem pressures caused by nutrient discharges. However the LCA research community has been relatively slow to resolve a methodology for *EP* that has a high degree of relevance for wastewater discharges around the world.

Early approaches tended to sum all fluxes of nitrogen (N) and phosphorus (P), comparing the two on the basis of the Redfield ratio – the standard approach to ascertaining the relative N and P uptake by waterborne algae (Redfield *et al.* 1963). However this ignored 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.

A number of different methods have grown from this, reflecting particular regional characteristics of relevance to Europe (Goedkoop *et al.* 2009), the United States (Bare *et al.* 2006; Helmes *et al.* 2012) and Japan (Hirosaki *et al.* 2002). Consideration of waterways sensitivity introduces the requirement for instream fate modelling, which has been handled in a variety of ways by these models.

There is no equivalent model available to inform the application of LCA across Australia. Australian LCA practitioners are therefore faced with the choice of how best to apply Eutrophication Modelling on a case-by-case basis. The two most common approaches are to (a) assume that all N and P discharges matter; or (b) assume that only N matters for marine waterways, and only P matters for freshwater systems.

Discharges of both N and P are considered to pose risks to the ecological health of SEQ waterways, whether the emissions are entering coastal freshwater streams, estuaries, or directly into Moreton Bay (Abal *et al.* 2005). We therefore chose to account for all emissions of N and P (using the Redfield Ratio for N vs. P equivalency) in the *EP* model applied in this study. Impact factors are summarised in Lane *et al* (2011). While this approach will have less relevance in other parts of the country, difficulties in disaggregating the life-cycle inventory data precluded the use of a more sophisticated approach.

The *EP* model used in this project also accounts for the primary oxidation effects associated with emissions of organic matter, and the nitrification of ammonia. This is consistent with the approach favoured by some in the Australian LCA community (Grant *et al.* 2008; Foley 2009; Bengtsson *et al.* 2010). Others have suggested that this is an inappropriate hybridisation, given that oxygen depletion is only one of the three possible pathways from nutrient discharge to ecological degradation (ECJRC 2010). However, we feel that this is a worthwhile addition for LCA application in the Australian urban water sector, given the strong focus around Australia on minimising BOD and NH_4^+ inputs to waterways.

Along with those issues noted above, there are a number of additional shortcomings to the LCA modelling approach adopted here. Analysis based on annual emission loads for total N and total P will overlook any influence of nutrient speciation (organic vs. mineral), nutrient form (dissolved vs. particulate), and the timing and intensity of emissions into ephemeral or fluctuating systems. There are no precedents for the inclusion of issues such as these in international LCA eutrophication models.

A more substantial review of the most appropriate metrics for use in SEQ is warranted, given the importance of nutrient discharges and eutrophication risks to the local urban planning regime. Doing so would be entirely consistent with the general trend towards spatial disaggregation of LCA impact models (Finnveden *et al.* 2009), and could form the basis for developing a framework that can be applied across the rest of Australia.

4.2. Ecotoxicity Potential

As discussed in our companion report on wastewater recycling (Lane *et al.* 2012a), LCA toxicity models could play a valuable role in providing a broader perspective on concerns about toxic chemicals (organic and metal) in urban wastewater streams.

Unfortunately, the fidelity of the available models is not sufficient to provide meaningful analysis in the urban water system context. Ecotoxicity modelling is therefore excluded from the majority of the analysis in this report.

The shortcomings of the available models for aquatic ecotoxicity are discussed in detail in a companion report on wastewater recycling (Lane *et al.* 2012a). Coverage of terrestrial ecotoxicity modelling is provided in Chapter 6 of this report.

4.3. Minerals Depletion

There are a number of different metrics used for LCA analysis of mineral resource depletion, but no consensus in the research community on the best approach to be taken. The most commonly used approaches can be divided into three groups:

- The first group of metrics are based on physical resource availability, and is the recommended approach from a recent study into LCIA models for European use (ECJRC 2011). However, more recent workshops involving LCA researchers and minerals industry stakeholders, concluded that resource availability is a poor proxy for the future significance of minerals depletion (Vieira *et al.* 2011).
- A second class of metrics are thermodynamically based, with the advantage that they rely solely on fundamental, robust physico-chemical data. However, thermodynamic based metrics provide no perspective on the implications of resource scarcity, hence can only be used for a limited range of purposes.
- The third approach, as applied in the ReCiPe models, is to compare different minerals on the basis of their cost to society if the production of each is increased by some marginal amount. Current UN funded research is also developing metrics using this same concept, albeit in a modified form (Vieira *et al.* 2012).

The starting point for the analysis in this project is the *Minerals Depletion* impact model provided in ReCiPe (Goedkoop *et al.* 2009). This is the only available model that avoids the limitations identified above, and maintains some consistency with the latest research direction on this topic.

Unfortunately, the majority of LCA *Minerals Depletion* models do not provide impact factors for phosphorus, and therefore cannot adequately account for the implications of phosphate rock depletion. This would seem a critical gap, given (a) international concerns over potential supply-demand imbalances that could affect world food production (Cordell *et al.* 2009); and (b) urban wastewater systems have been identified as one of the major sources of phosphorus that could be recovered from the anthropogenic system (Cordell *et al.* 2011).

In Chapter 6, we explore the implications of using different impact models so that the significance of phosphorus recovery can be assessed. As a part of this, we estimated an impact factor for phosphate resources in a manner consistent with the *Minerals Depletion* impact model of ReCiPe. This was calculated using publicly available phosphate rock mining data from the United States Geological Survey (Jasinski 2009). However, this impact factor can be considered as an interim value only. While the USGS databases did yield sufficient information, the most suitable data necessary for this task is not publicly available. Because of this, the quality of our estimated impact factor cannot be ascertained.

5. CASE STUDY - ADVANCED STP NUTRIENT REMOVAL

A major plant upgrade has recently been implemented for the South Caboolture STP, in part to cater for future population growth in its sewage catchment, and in part to improve the level of nutrient removal that can be achieved. Lower effluent nutrient levels will remove the need for downstream effluent polishing in the South Caboolture Water Recycling Plant (WRP). In the longer term, it will allow the low effluent nutrient levels to be maintained, as the population grows.

It is by no means a given that reducing nutrient discharge loads will deliver an overall reduction in the life-cycle environmental burden of the urban wastewater system. A previous study concluded that local aquatic eutrophication would need to be considered ~3 times more important than all other environmental issues for this to be the case (Foley 2009).

In SEQ, improvements in STP nutrient removal performance are largely being driven by the regional water quality targets set through the Healthy Waterways planning process. This confirms that the SEQ policy community <u>does</u> place a very high priority on reducing local eutrophication impacts. However, it could be argued that the evolution of this management paradigm did not reflect any informed choices about the prioritisation of local vs. external/global environmental impacts. Rather, it simply represents a particularly successful example of a local environmental problem being adopted into regional policy.

The analysis of Foley (2009) was based on a hypothetical set of STPs, however it does provide a framework that could be used to critique real-life STP upgrades that are being planned or delivered in SEQ. In this chapter, we use some quasi-hypothetical scenarios for the South Caboolture STP to illustrate how such a framework might be applied.

It is not the goal of this chapter to provide definitive comment on the actual infrastructure program implemented at the plant in question. Rather, the aim is to demonstrate how LCA can be used in practice to assist local infrastructure planning (Objective 1 of this report). A breakdown of the case study results is provided, but only to illustrate the key aspects of the wastewater system that influence the LCA results (Objective 2).

Detailed GHG analysis for the STP upgrade scenarios is also provided, in order to further the contribution to Objectives 2 and 3 of this report. This provides a practical demonstration of the fugitive gas estimation techniques, challenges and research gaps described in Chapter 2. It also puts the fugitive gases into the context of other life-cycle GHG emissions.

5.1. Case Study Overview

5.1.1. System Boundary and Functional Unit

The analysis in this chapter is based on simplified models for the STP operations, before and after the upgrade. These were developed using a combination of plant operations data and insight from the process engineering designs for the STP upgrade.

The real-life infrastructure upgrade implemented at the South Caboolture STP was designed to both increase its treatment capacity, and improve its nutrient removal performance. As the goal of this chapter is to explore the implications of an increased level of STP nutrient removal, the scenarios developed for this analysis differed from reality in two ways:

- Firstly, the scenarios were designed to exclude changes associated specifically with the capacity component of the infrastructure upgrade. To do this, the sewage characteristics (inflows, nutrient and COD concentrations), and the locations of wastewater, grit and biosolids disposal, were kept constant across both scenarios. The modelling only included those changes in auxillary inflows (power and chemicals use) and byproduct flows (grit, biosolids) that were required to deliver the change in nutrient removal.
- Secondly, the scenarios developed for this chapter ignored the presence of the South Caboolture WRP. This was done to give the findings greater relevance for situations where downstream nutrient removal capacity does not exist.

The scenarios were compared on the basis of the following Functional Unit:

The collection and treatment of 1 year's sewage from the existing South Caboolture STP catchment, assuming all treated wastewater is discharged to the Caboolture River, and assuming that all solids byproducts are disposed to landfill.

5.1.2. Data Sources – General Operations

Where available, operational inventories were based on the empirical STP models developed from the plant operations data (Table 5-1). The model for the pre-upgrade scenario was calibrated against process operations data provided by Unity Water. For the post-upgrade scenario, the model reflects a combination of this historical data, and design information from the current upgrade project.

No operational data specific to the South Caboolture sewer network was collected. Instead we used estimates for sewer pumping energy taken from a previous study at the Gold Coast study (Lane *et al.* 2011). The possibility of sewer chemicals dosing was excluded from this analysis.

Table 5-1: Summary of the two scenario characteristics (pre- and post-upgrade of the STP nutrient removal capability), showing key operating parameters and the relevant data sources.

		Scenario (1) *STP Pre-Upgrade*	Scenario (2) *STP Post-Upgrade*
STP Throughput		ADWF = 8.3 ML/d; AWWF = 10.8 ML/d (historical plant data)	
effluent distribution		100% to Caboolture River (hypothetical)	
Sewage Concentrations		TKN=56 mg/L; TP=11 mg/L; COD=580 mg/L (historical plant data)	
Effluent Dry Weather Concentrations		TN=7.2 mg/L; TP=7.1 mg/L COD=35 mg/L (historical plant data)	TN=2.5 mg/L; TP=0.3 mg/L COD=35 mg/L (design plans for STP upgrade)
N, P, C Balances		(based on STP design expertise and emission factors for fugitive gases)	
Biosolids Production		1.8 dry tonnes/d (historical plant data)	2.4 dry tonnes/d (modelled)
		trucked 150km to landfill (historical plant data)	
Power Use	**Sewage Networks and STP Feed Pumps**	2.6 MWh/d	
	STP Treatment	5.8 MWh/d (historical plant data)	7.4 MWh/d (model based on combination of historical and design data)
Chemicals Use	**Sewage Networks**	--	
	STP Treatment	40 kg/d (historical plant data)	4500 kg/d (model based on design plans for STP upgrade)
Misc		primary screenings; fuel use (historical plant data)	

5.1.3. Data Sources – Fugitive Greenhouse Gas Emissions

In all but one case, the analysis in this chapter uses the emission estimation approaches described in Chapter 2 (Table 2-1). For estimating N_2O generated from the nitrification/denitrification steps in the STP treatment train, a modified approach was used. This was required to cater for uncertainty over how the N_2O emissions intensity might change as a result of the increased nutrient removal.

In the absence of more focussed monitoring or process modelling for the South Caboolture STP, simplified high- and low-bound estimates were used to explore a range of possible changes in N_2O emission rates. To do this, we first estimated the total N_2O emission flux for the pre-upgrade scenario (1) using the methodology described in Chapter 2. We then determined two possible extremes in the effect of the upgrade on N_2O emissions (Table 5-2).

The first possibility that was considered (scenario 2a) assumed that 100% of the N_2O emissions in the pre-upgrade scenario are caused by the denitrification step, and that the N_2O emissions would increase proportionately with an increase in the amount of N denitrified. The second possibility (scenario 2b) assumed that nitrification is the sole source of N_2O; and that the upgrade would result in no change to N_2O emission rates from the STP.

For the analysis of GHG profiles (section 5.1), an additional set of results was generated using the conventional NGERS approach (DCCEE 2011). The emission factors used for the NGERS-based calculations are also summarised in Table 2-1.

Table 5-2: Alternatives used for sensitivity analysis on the changes to N_2O generation in the STP.

	Pre-Upgrade	Post-Upgrade	
Scenario	**(1)**	**(2a)**	**(2b)**
Assumed source of N_2O generation (pre- and post-upgrade)	not specified	100% from denitrification	100% from nitrification
Change in control variable	--	10% increase in the quantity of N denitrified	no change in the amount of nitrification
Change in N_2O flux	--	0.35 kg-N_2O/d	0 g-N_2O/d
N_2O flux from the STP	3.45 kg-N_2O/d[1]	3.8 kg-N_2O/d	3.45 kg-N_2O/d

[1] Based on an emission factor of 8.5 g-N_2O per kg-ΔN, where $\Delta N = TKN_{INFLUENT} - TN_{EFFLUENT}$

5.1.4. Data Sources - Construction

For developing an indicative GHG breakdown over the entire wastewater system, the pre-upgrade scenario model included estimates for the construction of sewer network and STP infrastructure. STP construction inventories were taken from those of the closest technology configuration considered by Foley *et al.* (2010a). Construction inventories for the sewer network assumed the same distributions of pipe size and materials type in the data collected for previous analysis of the Gold Coast urban water system (Lane *et al.* 2011). All construction materials inventories were pro-rated on the basis of their expected lifetimes.

Future infrastructure disposal in the end-of-life phase was not included in the analysis, as this is typically found to have negligible effect on the LCA results for urban wastewater systems (Foley *et al.* 2010a).

Construction inventories were not included in the modelling of changes associated with the upgrade. No changes were required for the sewer network inventories, since the post-upgrade scenario did not incorporate any increase in the catchment EP. No data was collected on the additional infrastructure required for the STP upgrade.

5.1.5. Data Sources – Background Inventories

Data for power generation (assumed to be from the East Australian Grid), and the manufacture of chemicals and construction materials, were taken from other sources. The Simapro Australasian database (Grant 2012) was used in most cases, otherwise the chosen datasets were based on those from Ecoinvent (Frischknecht *et al.* 2007). Further discussion on the collection and use of background data is provided in a companion report (Lane *et al.* 2012c).

5.2. GHG Profile of the Wastewater System

Power and Chemicals Use

Table 5-3: GHG footprint for the South Caboolture STP prior to the nutrient removal upgrade.

	GHG footprint (t-CO2e / y)
total - scope 1 (fugitives*)	**830 (20%)**
N2O - STP	350 (9%)
N2O - biosolids disposal	82 (2%)
N2O - WW discharge	79 (2%)
CH4 - sewer	395 (10%)
CH4 - STP	0 (0%)
CH4 - biosolids disposal	372 (9%)
CO2 - STP	72 (2%)
CO2 - biosolids disposal	-650 (-16%)
other	130 (3%)
total - scope 2 (power)	**2,820 (70%)**
sewer pumping	468 (12%)
STP treatment	2,352 (58%)
WW discharge	0 (0%)
total-scope 3	**380 (9%)**
chemicals - sewer	0 (0%)
chemicals - STP	6 (0%)
chemicals - WW discharge	16 (0%)
biosolids - transport	53 (1%)
construction - networks	178 (4%)
construction - STP	129 (3%)
other	20 (0%)
total	**4,050 (100%)**

** using the science-based approach to estimating fugitive emissions.*

Table 5-4: Key changes to the GHG footprint of the South Caboolture STP as a result of the nutrient removal upgrade, showing the changes as a % of the total GHG footprint for the pre-upgrade scenario.

The results are calculated using (a) science-based and (b) NGERS-based estimates for fugitive gas emissions.

(a)	GHG footprint (t-CO_2e / y)			
	pre-upgrade	post-upgrade	change to pre-upgrade total	
total - scope 1 (fugitives*)	**830**	**810 / 760**	**-20 / -70**	**-1% / -2%**
N2O - STP	350	403 / 350	+53 / 0	1% / 0%
N2O - biosolids disposal	82	109	27	+1%
N2O - WW discharge	79	28	-52	-1%
CH4 - biosolids disposal	372	497	125	+3%
CO2 - STP	72	117	45	+1%
CO2 - biosolids disposal	-650	-874	-224	-6%
other	524	524	0	
total - scope 2 (power)	**2,820**	**3,360**	**540**	**+13%**
STP treatment	2,352	2,889	536	+13%
other	468	468	0	
total - scope 3	**380**	**1,670**	**1,290**	**+32%**
chemicals - STP	6	1,278	1,272	+31%
biosolids - transport	53	70	18	+0%
other	321	321	0	
other	20	130	110	+3%
total	**4,050**	**5,970 / 5,910**	**1,910 / 1,860**	**+47% / +46%**

$^{\Delta}$ using the science-based approach to estimating fugitive emissions.

(b)	GHG footprint (t-CO_2e / y)			
	pre-upgrade	post-upgrade	change to pre-upgrade total	
total - scope 1 (fugitives#)	**810**	**1,000**	**190**	**+5%**
N2O - STP	437	502	65	+2%
N2O - biosolids disposal	0	0	0	
N2O - WW discharge	0	0	0	
CH4 - biosolids disposal	372	497	125	+3%
CO2 - STP	0	0	0	
CO2 - biosolids disposal	0	0	0	
other	0	0	0	
total - scope 2 (power)	2,820	3,360	540	+13%
total - scope 3	380	1,670	1,290	+32%
total#	**4,010**	**6,030**	**2,020**	**+50%**

 # recalculated using the NGERS approach to estimating fugitive emissions

Power use was the dominant contributor (70%) to the total GWP of the pre-upgrade wastewater system at South Caboolture. In particular, power use for treatment processes contribute more than half (58%) of the total GWP (Table 5-3). If considered on a flow-specific basis (700 kWh/ML), the pre-upgrade estimate for treatment power use is close to the mean (720 kWh/ML) of data collected for 21 treatment plants in SEQ (de Haas *et al.* 2009).

The nutrient removal upgrade is estimated to increase the energy requirement for treatment by 20%, or 536 t-CO_2e/y (Table 5-4). Despite this, the treatment power use (2889 t-CO_2e/y) contributes less than half of the overall GWP (~5910 t-CO_2e/y) of the post-upgrade scenario.

The primary reason for the divergence between energy use, and the overall GHG footprint, is the intensive alum dosing required for the additional phosphorus removal capacity. This is responsible for the single biggest increase (31%) in overall GWP associated with the BNR upgrade (Table 5-4). Prior to the upgrade, chemicals use at the South Caboolture STP was relatively minor.

The significant role of GHG 'embedded' in chemicals use suggests that decision making based on the compulsory components (Scope 1 and Scope 2 emissions) of the NGERS protocol might fail to reveal substantial GHG burdens being introduced to the wastewater system. If the future cost of chemicals were to increase as a result of economy wide pricing pressures (energy cost increases, or carbon pricing), then this could translate to an unforeseen cost burden for the WWT plant operator. This issue has already been investigated by Sydney Water, which estimates that carbon pricing will lead to substantial pass-through costs from their chemical suppliers (Kenway *et al.* 2008; Sydney Water 2011).

More accurate estimates of the GHG intensity of chemicals supply might therefore be beneficial. This is the focus an ongoing research project at UNSW, which aims to generate high quality manufacturing inventories for the most commonly used chemicals in the Australian water industry (Gaitan pers comms). However, it is not clear when that information will be made available for public use. Accessing commercially sensitive data such as this can be a significant challenge for LCA related analysis.

Fugitive Gas Emissions

Fugitive gas emissions make the other major contribution (20%) to the overall GWP of the pre-upgrade scenario (Table 5-3). The three most notable sources are associated with sewer CH_4 generation, N_2O emissions from the STP, and biosolids carbon. In all three cases, these estimates are subject to large uncertainties. Furthermore, each of these three fugitive gas sources is likely to vary substantially across different sites. For N_2O and sewer generated CH_4, this suggests that some wastewater treatment plants may eventually be exposed to substantially bigger GHG burdens than are accounted for under the current reporting framework.

The choice of assumptions used for estimating the change in N_2O emissions does not have a significant effect on the scale of the overall GWP increase caused by the nutrient removal upgrade (Table 5-4a). Even using the NGERS accounting approach (Table 5-4b), the highest effective emission factor for STP N_2O generation that was considered in this study, the increase in N_2O (65 t-CO_2e/y) is much smaller than those associated with power use (536 t-CO_2e/y) or chemicals use (1272 t-CO_2e/y). Furthermore, the increase in N_2O generation at the STP is offset by the decrease in generation from the wastewater receiving environment. It is only because of the small increase in N_2O from biosolids disposal, as a result of the higher rate of biosolids generation, that the overall N_2O flux increases.

These results illustrate that predicting the GHG implications of changes in nutrient removal will be dependent on realistic estimates for the offsite N_2O emissions associated with wastewater and biosolids disposal. Along with better characterisation of STP emissions, further research into N_2O emissions from these offsite points would also be valuable.

In total, the change in overall GWP associated with changes in fugitive gas emissions is minor and negative (-20 to -70 t-CO_2e/y), if using the science based set of emission factors (Table 5-4a). In contrast, using the NGERS estimation approaches would indicate that the GWP increases by 190 t-CO_2e/y, or a 5% increase in the overall GWP for the STP (Table 5-4b).

This highlights the problem with using the NGERS protocol as the basis for assessing fugitive GHG emissions in planning studies. Such planning studies are essentially forecasting exercises, yet NGERS emission factors represent historical information that is rapidly being superseded. Water utilities in the future will likely need to account for a much greater range of fugitive emission pathways than is currently the case. Using the NGERS approach, in current-day decision making, is not likely to provide useful guidance on changes to the future GHG footprint for a water utility.

5.3. A Broader Set of Environmental Tradeoffs

Identifying the Tradeoffs

Results for Scenario (1) and Scenario (2b) were also calculated using the full set of LCA impact categories, so as to consider a range of other environmental externalities.

The large decrease in *Eutrophication Potential* (-88%) comes at the expense of substantial increases for a number of the other impact categories (Table 5-5). These downsides (increased impact results) represent the tradeoffs associated with the advanced nutrient removal processes that have been adopted. Most of the downsides are associated with the increased power and chemicals supply required for the additional nutrient removal, although the different impact categories are not uniformly sensitive to each (Table 5-6).

The increased estimate for N_2O flux from the secondary treatment process caused a 9% increase in overall *Ozone Depletion Potential*, however this is a relatively small change compared to those in most of the other impact categories. As discussed in the previous section, changes in fugitive gas fluxes (CO_2, CH_4, N_2O) made a relatively small contribution to the overall increase in *Global Warming Potential*.

Clearly, our results suggest that changes in power and chemicals use, not fugitive gas fluxes, are likely to be the most substantial cause of life-cycle environmental burdens associated with the upgrade of the South Caboolture STP. However, caution should be taken in extrapolating this finding to other contexts, as it is dependent to a large degree on the nature of this particular case study. Firstly, because the results could be very different, if different assumptions on fugitive gas emissions were used. Secondly, because previous analysis (Lane *et al.* 2011) showed that life-cycle environmental burdens can vary substantially from chemical to chemical.

Table 5-5: Change in LCA impact results due to the change from scenario (1) to scenario (2b).

		pre-upgrade	post-upgrade	increase	
Eutrophic'n Potential	(t PO4-e/y)	102	12	-90	(-88%)
Freshwater Extraction	(ML/y)	10	12	+2	(+20%)
Global Warming	(kt CO2-e/y)	4	6	+2	(+49%)
Ozone Depletion	(kg CFC11-e/y)	34	37	+3	(+9%)
Fossil Fuel Depletion	(t oil-e/y)	840	1,297	+457	(+54%)
Minerals Depletion	(t Fe-e/y)	113	116	+3	(+3%)
Human Toxicity	(t 1,4-DCB-e/y)	57	83	+26	(+45%)
Photochem oxidants	(t NMVOC/y)	15	26	+11	(+72%)
Particulates formation	(t PM10-e/y)	7	13	+7	(+105%)

Table 5-6: Contributing factors to the difference in LCA impact results between scenario (1) and scenario (2b).

	overall	power	chemicals	fugitive gases	other
Eutrophic'n Potential	-88%				-89%
Freshwater Extraction	+20%		+15%		
Global Warming	+49%	+13%	+31%		+3%
Ozone Depletion	+9%	+2%		+5%	
Fossil Fuel Depletion	+54%	+16%	+29%		+9%
Minerals Depletion	+3%		+2%		
Human Toxicity	+45%	+7%	+34%		+3%
Photochem oxidants	+72%	+16%	+49%		+4%
Particulates formation	+105%	+14%	+79%	+4%	+2%

*** mismatches are a result of rounding errors*

The Need for Perspective

Prioritising across those tradeoffs identified in Table 5-5 would likely be a challenging exercise for urban water system managers. They would be unfamiliar with most of the environmental issues introduced by the LCA methodology, and would therefore require some perspective on the importance of each of the different impact category results.

To help provide this perspective, previous studies on nutrient removal have normalised (benchmarked) wastewater treatment case study results against an estimate of the total life-cycle burden associated with the Australian economy (de Haas *et al.* 2008; Foley 2009). That benchmark dataset includes an estimate of the total emissions (to air, water and soil) and total resource extractions (fossil fuels, minerals, water) for all activities occurring in Australia.

For analysis of an existing STP, this approach to benchmarking gives an indication of how much the plant would contribute to the overall Australian life-cycle environmental burden associated with each impact category. For analysis of a system change, such as the upgrade of an STP, it provides an indication of how much that action would change the overall impact burden of the Australian economy. In either case, comparing the tradeoffs on this basis provides some insight into how they might be prioritised from a broader social perspective.

The relevance of that Australian perspective is contingent on the notion that the overall objective for society is to eliminate its entire environmental burden. In practice, however, there will never be sufficient resources available for complete environmental impact mitigation. As a result, decision makers at all levels of society must prioritise those environmental issues which are felt to warrant the greatest attention in the short term.

For the multi-criteria approach used in conventional LCA, the simple way to implement this prioritisation approach into decision making is to weight each of the criteria under consideration. Foley (2009) used this approach to illustrate how sensitive the nutrient removal debate might be to different impact prioritisation perspectives. His analysis suggests that mitigation of local eutrophication would have to be considered three times more important than for all other life-cycle impact categories, if the trend to increased STP nutrient removal is to be justified.

Unfortunately, it is difficult to ascertain the credence of Foley's (2009) quantitative conclusions, due to some fundamental constraints with interpreting LCA results that are normalised in this fashion. One such constraint is the uncertain quality of the Australian benchmark dataset, both in terms of data completeness and data precision. The potential influence of gaps and uncertainties in the benchmark dataset will not be uniform across different LCA impact categories, nor will it be transparent to users of LCA normalisation (Lane *et al.* 2012c). As a consequence, there is significant potential for unrecognised bias to affect the interpretation of normalised results (Heijungs *et al.* 2007).

For urban water systems analysis, an alternative approach is to benchmark the case study results against an estimate of the total estimated life-cycle impact associated with the entire urban water system (Lane *et al.* 2011; Lane *et al.* 2012c). The benchmarks used in those examples were based on detailed data for the urban water systems; hence the benchmark dataset is relatively well understood and the potential for hidden bias is therefore reduced. A second advantage is that a benchmark focussed directly on the environmental performance of the urban water system, might be considered more relevant to the decision making of local water utilities.

However, it is important to recognise that there is no 'perfect' way to normalise or benchmark LCA results (Lane *et al.* 2012c). All possible approaches have their limitations. In general terms, reducing the spatial scale of the benchmark will increase the data quality and increase the institutional relevance. The compromise is that the perspectives gained will be more insular. Broader scale benchmarks (such as that used for the Australian economy) are more likely to suffer from data constraints, but do provide more holistic perspectives that are more closely matched to the scope of life-cycle analysis.

Regardless of the limitations inherent in the LCA normalisation step, the framework used by Foley (2009) can still be used to explore the bigger picture implications of pursuing increased levels of STP nutrient removal. The fundamental basis of the Foley (2009) approach is to ascertain the sensitivity of conclusions to different approaches to impact prioritisation. Applying this framework to specific nutrient removal case studies could therefore provide a useful perspective on the tradeoffs involved.

Prioritising the tradeoffs

To demonstrate how the normalisation approach could be used to assess the tradeoffs associated with a nutrient removal upgrade, we normalised the change in LCA results (from Table 5-5) against an Australian benchmark. Table 5-7 shows that the significance of the EP change is substantial enough to exceed the total of the normalised changes for all other impact categories.

This implies that, if all impacts were to be considered equally important, the net effect of the STP upgrade is a net reduction in overall (normalised) impact. The overall tradeoffs balance could only indicate a net increase in environmental burden if local eutrophication was considered less of a priority than the other life-cycle environmental impact categories.

Clearly this is not the case in practice. As described in the introduction, the reduction of nutrient inputs to Moreton Bay is considered one of the highest priority environmental drivers for decision making under the TWCMP process. Our results indicate, therefore, that the overall life-cycle implications of the nutrient removal upgrade at the South Caboolture STP could be considered a positive development.

This is the opposite conclusion to that drawn by Foley (2009) and de Haas (2008). As discussed above, those studies suggested that local eutrophication would have to be valued disproportionately highly for the overall trend towards increased nutrient removal to be justified.

One possible explanation for this discrepancy is that we used a different set of LCA impact models to those in the older studies. More detailed analysis would be required to confirm whether or not the choice of LCA models would affect the conclusions drawn for this particular case study.

It is also possible that the discrepancy is because the analysis in this report covers a single, isolated case study. It could simply be that this is one example where the implications buck the trend that was unearthed in the previous studies.

Table 5-7: Normalised results for the change from scenario (1) to scenario (2b).

			Normalised against Australian impact estimates		
		case-study change	benchmark	normalised change	% change to benchmark
Eutrophic'n Potential	**(t PO4-e/y)**	**-90**	**254,186**	**-354 E-6**	**-0.0354%**
Freshwater Extraction	(ML/y)	+2	0	0.0 E-6	0.0000%
Global Warming	(kt CO2-e/y)	+2	580,611	+3.4 E-6	+0.0003%
Ozone Depletion	(kg CFC11-e/y)	+3	1,410,700	+2.1 E-6	+0.0002%
Fossil Fuel Depletion	(t oil-e/y)	+457	267,702,020	+1.7 E-6	+0.0002%
Minerals Depletion	(t Fe-e/y)	+3	449,451,600	+0.0 E-6	+0.0000%
Human Toxicity	(t 1,4-DCB-e/y)	+26	32,894,310	+0.8 E-6	+0.0001%
Photochem oxidants	(t NMVOC/y)	+11	1,667,174	+6.6 E-6	+0.0007%
Particulates formation	(t PM10-e/y)	+7	1,725,226	+4.1 E-6	+0.0004%
total of normalised results for other impact categories (excluding *Eutrophication Potential*)				**+28 E-6**	**+0.0028%**

5.4. Conclusions – LCA and Wastewater Nutrient Removal

Wastewater Systems Planning requires More Robust Greenhouse Gas Analysis

The NGERS accounting system forms the basis for the majority of GHG auditing in Australia's urban water industry. NGERS encourages a focus on Scope 1 (direct gas emissions) and Scope 2 (power use) emissions, with reporting of Scope 3 emissions (embedded in chemicals and materials supply) being voluntary. The NGERS system is primarily designed to audit the GHG intensity of various industry sectors and individual businesses.

However, for water utilities wishing to minimise their future GHG footprints, the NGERS Scope 1 and 2 accounting frameworks are unlikely to provide an adequate basis for planning decisions. For capital expenditure to deliver long term and robust benefits, decision making processes should consider the potential future exposure of utilities to environmental and market pressures, rather than relying on the status quo frameworks for carbon accounting.

The adoption of chemicals intensive processing steps could expose wastewater system operators to substantial supply chain cost increases, as energy and carbon pricing systems change into the future. Including Scope 3 accounting into decision making frameworks would therefore help to identify, and potentially reduce, the chances of this occurring. Unfortunately, the quality of available information on the GHG intensity of chemicals supply is uncertain. Improvements to this data would be beneficial, as would further development of techniques to ascertain the future risk of supply chain cost increases. Utilities in other parts of Australia have demonstrated how supply-chain price risk analysis can be proactively inform their service pricing structures, and SEQ utilities might benefit from adopting a similar approach.

Using the NGERS protocols could also substantially misrepresent the fugitive gas (Scope 1) implications of changes in STP operating conditions. For the case study analysed in this report, the use of science-based emission factors indicated a negligible GHG burden associated with a substantial increase in STP nutrient removal. In contrast, the NGERS approach would penalise the STP operators for the increased denitrification and biosolids disposal required to meet the desired effluent nutrient concentrations.

Unfortunately, there also major limitations with the information available to support science-based fugitive gas estimates. While there have been substantial recent improvements in the fundamental understanding of wastewater related CH_4, CO_2 and N_2O emissions, the majority of that research is focussed on ascertaining overall levels of emissions for existing systems. For the majority of the relevant pathways, there remain major barriers to applying this insight in analysis that is focussed on *changes* in wastewater system operations.

Wastewater Systems Planning requires More Robust Environmental Tradeoffs Analysis

Changes in energy and chemicals use dominated not just the GHG results, but also most of the other life-cycle environmental burdens associated with the nutrient removal upgrade considered in this report. However, it is not a given that GHG accounting will provide a robust proxy for more broad spectrum tradeoffs analysis in wastewater system planning. Firstly, fugitive gas emissions could conceivably be far more important under other conditions. Secondly, the different LCA impact categories are not uniformly sensitive to power use, fugitive gas emissions, nor the variety of chemicals products used by the wastewater industry.

The consideration of a wider range of environmental issues will introduce additional challenges for water system planners, as it increases the complexity of tradeoffs analysis required in the decision making process. A framework does exist for comparing the significance of wastewater nutrient removal, and the local water quality benefits that it can deliver, with the disparate, offsite impacts that are generated as a result. However, methodological improvements are required to make this framework operational.

Use of this framework would make it directly possible for water utilities to identify and articulate the benefits (or otherwise) of pursuing individual projects targeted at enhanced nutrient removal. However, a more systematic investigation is required to inform higher-level policy debates on the justification for pursing enhanced STP nutrient removal in SEQ. There has already been sufficient STP case study data collected through other projects (de Haas *et al.* 2009; Foley 2009) to support such analysis. Coupling this data with recent and ongoing advances in LCA methodology, would provide powerful comment on the bigger picture implications of the current paradigm.

6. CASE STUDY - BIOSOLIDS DISPOSAL

Unitywater's recent Biosolids Management Strategy recommended that the South Caboolture STP adopt agricultural reuse as the preferred biosolids disposal strategy, replacing the traditional practice of using the solids as fill material. This policy direction reflects the financial and strategic risk associated with continuing to rely on the availability of landfill to take the biosolids material. It is also consistent with the commonplace notion that agricultural biosolids reuse delivers broad environmental benefits.

Based on previous analysis of the Gold Coast urban water system (Lane *et al.* 2011) suggests that biosolids options analysis was not subjected to a robust assessment of the bigger picture, life-cycle environmental implications. In that study, agricultural reuse of biosolids was identified as a major contributor to the environmental burden of the overall urban water system. However, it also highlighted a number of methodological problems that could constrain the utility of LCA for biosolids systems analysis.

For urban centres with an interest in agricultural biosolids reuse, there exists a second imperative to understand the implications of analysing biosolids disposal options under the LCA framework. The LCA methodology is rapidly gaining traction as a tool for assessing Australian agricultural management practices. This means that biosolids reuse will come under increasing scrutiny from outside the water industry.

The biosolids case study in this report explores the benefits and challenges involved in using LCA to assess different biosolids disposal options (Objectives 1 and 2 of this report). A number of key gaps are identified that should be the focus of further data collection or methodological improvement (Objective 2). Examples are provided for estimating key assumptions of relevance to the analysis of biosolids disposal options (Objective 3).

While we use here a South Caboolture based case study, the analysis in this report is largely generalist in nature. A detailed critique of the options specifically available to Unitywater was beyond the scope of this project, and would require the use of more locally specific data on the location and nature of agricultural reuse opportunities. This analysis also focuses strictly on the disposal pathways, excluding any additional biosolids stabilisation processing that might be required to make the agricultural reuse possible.

6.1. Case Study Overview

6.1.1. System Boundary

Biosolids use as Fill Material

This scenario assumes that the biosolids are used as fill material at either landfill sites, or former mine sites, and then is subsequently buried (Figure 6-1a). It also assumes that no onsite physical infrastructure would be required for intermediate storage of the biosolids.

Direct Biosolids Application to Agriculture

This scenario assumes that stabilisation-grade B biosolids are trucked from the STP directly to grain producing farms for use as a fertiliser replacement (Figure 6-1b). The biosolids are applied by tractor at a rate equivalent to 1 NLBAR[2], in accordance with the Queensland biosolids application guidelines (Qld EPA 2002). As with the landfill disposal scenario, no on-farm physical infrastructure was included in the analysis.

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[2] NLBAR = Nitrogen Limited Biosolids Application Rate.

Fertiliser and Crop Offsets

Agricultural biosolids reuse was given a credit for the avoided use of synthetic fertiliser (urea and DAP). The inventories for this offset included the manufacture and supply of the fertiliser to the field.

A credit was also given for crop yield increases attributed to the biosolids application. A number of recent Australian studies have found that biosolids amended fields achieve greater crop yields than comparable fields fertilised with conventional, synthetic products (e.g. Barry *et al.* 2006; Powell *et al.* 2012). It was assumed that this increased crop production would offset the need for an equivalent amount of grain produced elsewhere, whether that be on the same farm or at a different location.

Inventories for the offset crop production were measured to the farm gate, and included the harvesting operations. Harvesting of the additional grain from the biosolids-amended fields was not included in the inventories, on the premise that harvesting fuel inputs are determined by the area covered, not by the volume of produce extracted. 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.

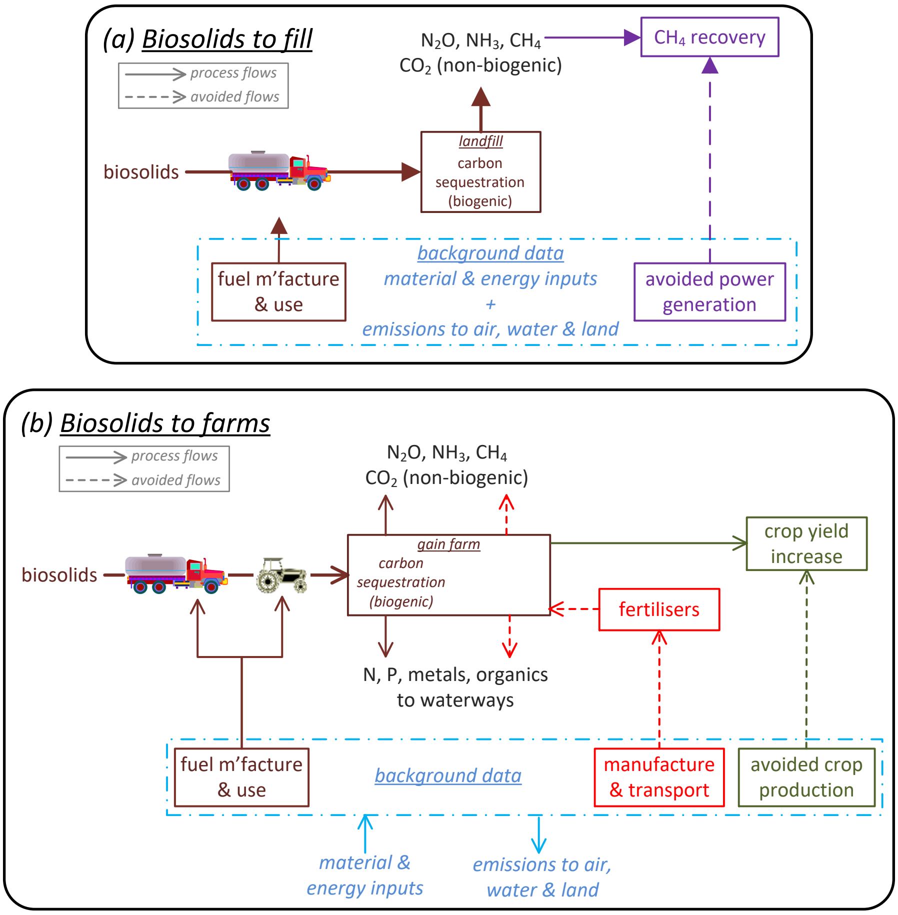

Figure 6-1: System boundary for the scenarios involving (a) disposal as fill material; and (b) direct application of biosolids to agricultural land. Solid lines represent flows that are attributed to the biosolids disposal. Dotted lines represent flows that are avoided by the use of biosolids, and allocated as credit to the biosolids scenario.

6.1.2. Data sources - biosolids generation

Data for biosolids generation rates and moisture content for the South Caboolture STP was provided by Unitywater, with the nutrient and carbon content modelled using full carbon, nitrogen and phosphorus balances across the sewage treatment plants in the MBRC region (Table 6-1).

Assumptions for the concentration in biosolids of trace metals and organics were based on data collected for four STPs at the Gold Coast (Lane *et al.* 2011). The hypothetical contaminant profile meets the requirements for Contaminant Grade C biosolids, which is the minimum standard required for agricultural biosolids reuse in Queensland and NSW (NSW EPA 1997; Qld EPA 2002).

Table 6-1: Biosolids data for the South Caboolture STP.

STP throughput (ADWF)	(ML/d)	8.3
biosolids generation	(t-ds/d)	2.4
biosolids composition		
- moisture	(w-w%)	86.5
- total nitrogen	(kg-N/t-ds)	60
- phosphorus (biologically bound)	(kg-P/t-ds)	14
- phosphorus (Fe-bound)	(kg-P/t-ds)	22
- carbon (biogenic)	(kg-C/t-ds)	277
- carbon (non-biogenic)	(kg-C/t-ds)	45

6.1.3. Data Sources - Biosolids Disposal as Fill Material

The biosolids trucking distance of 150km was taken from data collected for the South Caboolture STP in a previous study (de Haas *et al.* 2009). Information was not collected on the requirements for handling of the biosolids onsite (e.g. truck movements required to distribute the biosolids to different mine pits), hence this element of the system was excluded from the inventories.

Chapter 2 (Table 2-1) describes the approaches used to estimate the flux of fugitive gases, and the quantity of carbon sequestered, at the disposal site. We assumed that no groundwater leaching would occur from the disposal site.

Table 6-2: Assumptions used for modelling of biosolids disposal scenarios.

	Disposal as Fill Material	Application to Farmland	
Transport to site	150km (historical data)	150km (maintained for consistency)	
Onsite handling	nil	tractor use = 0.325 L of diesel per t-wet solids	(Foley *et al.* 2010a)
Fertiliser handling	n/a	nil	
Biosolids nutrient Bioavailability	n/a	50% of biosolids N is bioavailable	(Foley *et al.* 2010a)
		50% of biosolids P is bioavailable	(Bradford-Hartke *et al.* in prep)
		75% of bioavailable P will displace fertiliser addition over the long term	assumption for sensitivity analysis
Fertiliser nutrient Composition	n/a	Urea = 46.5% as N (w/w)	(Renouf 2012)
		DAP = 21.2% as N (w/w) N; 23.5% as P (w/w)	(Renouf 2012)
		fertiliser N and P are 100% bioavailable	
Fugitive gases and carbon sequestration	see Table 2-1	see Table 2-1	
Irrigation offsets	n/a	nil	
Crop yield increase	n/a	330 kg of grain / ha	(Barry *et al.* 2006)

6.1.4. Data Sources - Biosolids Use for Agriculture

Trucking distance of the biosolids to farms (150km) was kept consistent with that for the status quo scenario. An allowance was made for tractor use required to spread the biosolids across the crop fields.

Other possible material or energy inputs, that might be required for onsite biosolids handling, were excluded from the analysis.

Soil carbon and nutrient balances were calculated for the biosolids applied to fields. Assumptions on carbon sequestration and non-biogenic CO_2 emissions follow the rationales described in Chapter 2.1. Nutrient losses (to atmosphere and waterways) followed the assumptions used in a previous LCA study for the Gold Coast wastewater system (Lane *et al.* 2011).

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 would translate into crop yield improvements, and would therefore be accounted for through the incorporation of crop offsets.

Fertiliser Offsets

Unless noted otherwise, fertiliser offsets were modelled according to the process described in a previous LCA study on the Gold Coast urban water system (Lane *et al.* 2011). In summary:

- It was assumed that the biosolids nitrogen is only 50% bioavailable (Foley *et al.* 2010a), and therefore for any particular field receiving a biosolids application rate of 1 NLBAR, two units of biosolids nitrogen would be required to offset 1 unit of nitrogen in synthetic fertilisers.
- At 1 NLBAR application rates, the phosphorus loading from biosolids application is likely to be substantially greater than required for crop fertilisation under many conditions (Barry *et al.* 2006; Pritchard *et al.* 2007). While excess phosphorus is likely to accumulate in most Australian soils, and therefore potentially be available for crop uptake in subsequent years, it was assumed that changes in fertilisation practice would only match 75% of the bioavailable phosphorus in the biosolids (Lane *et al.* 2011). Since the South Caboolture STP employs a high degree of chemical phosphorus removal, it was assumed that only 50% of the biosolids phosphorus would be bioavailable. This follows the assumptions developed through a parallel study by Bradford-Hartke *et al.* (in prep).
- DAP offsets were calculated to reflect the crop P contribution from biosolids application, with urea offsets then calculated net of any reduced N input associated with reduced DAP usage.

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). Transport from the store to the farm was assumed to be over a distance of 50km. We assumed that the distribution of synthetic fertiliser addition would normally be incorporated into other field management activities. Hence, we did not allocate any change in farm tractor movements to the avoided synthetic fertiliser use.

Crop Offsets

Data on yield increases that occur as a result of biosolids reuse were taken from trials conducted on fields growing feed-sorghum in the Darling Downs (Barry *et al.* 2006).

There is limited publicly available LCA data on Australian cropping production systems, and it is difficult to ascertain the relevance of this data to the analysis required for biosolids reuse. However, it is expected that this will change in the near future. An ongoing project led by CSIRO is collating detailed LCA data across the spectrum of cropping types and locations that exist in Australia (Grant *et al.* 2012). As an interim approach, an inventory for conventional production of feed-sorghum in Australia was sourced from unpublished data (Renouf 2012).

6.1.5. Data Sources - Transport and Onsite Vehicle Use

Transport models for both scenarios were taken from the Australasian LCA inventory database (Grant 2012), using assumptions described in a parallel study (Bradford-Hartke *et al.* in prep). These models include inventory estimates for truck fuel use, exhaust emissions, and heavy metals runoff from tyre wear. They exclude the contribution from the product (biosolids or fertilisers) transport to the need for road maintenance, and the construction and maintenance of the vehicles themselves.

6.1.6. Data Sources - Urban Water System

The biosolids results are presented in the context of the full South Caboolture wastewater system modelled in the previous chapter of this report.

6.2. Implications of the Change in Disposal Approach

The first step for the analysis in this chapter was to identify the contribution of biosolids disposal to the *post-upgrade* South Caboolture sewage system modelled in Chapter 1.1. Two different scenarios were considered: firstly, assuming that all the STP biosolids are used as fill material (Table 6-3); and secondly, assuming that all the STP biosolids are applied directly to agricultural lands (Table 6-4).

Table 6-3: Selected LCA impact results for the wastewater component of Caboolture Scenario #1, assuming that all biosolids are used as fill material.

	Eutrophic'n Potential (t PO4-e/y)	Terrestrial Ecotox (t 1,4-DCB-e/y)	Global Warming (kt CO2-e/y)	Ozone Depletion (kg CFC11-e/y)	Fossil Fuel Depletion (t oil-e/y)	Minerals Depletion (t Fe-e/y)	Human Toxicity (t 1,4-DCB-e/y)	Particulates formation (t PM10-e/y)
Total	12	0.1	6	37	1297	117	83	13
biosolids disposal	0.2 (2%)	0 (0%)	-0.2 (-3%)	6.6 (18%)	32 (2%)	0 (0%)	3.2 (4%)	1.2 (9%)
sewerage + WWT	12	0.1	5.8	30	1158	6.3	64	12
infrastructure construction	0	0	0.3	0.1	106	110	16	0.7

Table 6-4: Selected LCA impact results for the wastewater component of Caboolture Scenario #1, assuming that all biosolids are used on agricultural fields.

	Eutrophic'n Potential (t PO4-e/y)	Terrestrial Ecotox (t 1,4-DCB-e/y)	Global Warming (kt CO2-e/y)	Ozone Depletion (kg CFC11-e/y)	Fossil Fuel Depletion (t oil-e/y)	Minerals Depletion (t Fe-e/y)	Human Toxicity (t 1,4-DCB-e/y)	Particulates formation (t PM10-e/y)
Total	16	3.6	6.2	37	1254	117	87	15
biosolids disposal	3.9 (24%)	3.5 (97%)	0.1 (2%)	7.2 (19%)	-11 (-1%)	-0.2 (0%)	7.6 (9%)	2.9 (19%)
sewerage + WWT	12	0.1	5.8	30	1158	6.3	64	12
infrastructure construction	0	0	0.3	0.1	106	110	16	0.7

A comparison across the two scenarios shows that the shift from landfill disposal to agricultural use causes notable increases in the overall wastewater related impacts for a number of the LCA impact categories. Two of the bigger increases (*Terrestrial Ecotoxicity Potential*; *Eutrophication Potential*) are discussed further in the following section.

Only two of the impact categories (*Fossil Fuel Depletion*; *Minerals Depletion*) showed a net decrease as a result of the change. However, in both cases, the effect on the overall life-cycle burden for the wastewater system was minor. The *Minerals Depletion* result is discussed further in the following section.

6.3. Impact Assessment Challenges for Biosolids Analysis

6.3.1. Ecotoxicity

The biggest change between the landfill (Table 6-3) and agricultural (Table 6-4) disposal pathways was for the *Terrestrial Ecotoxicity Potential* results, which increased by more than 30 times.

Trace metals in the biosolids, particularly Copper (Cu), Zinc (Zn) and Mercury (Hg), are the dominant cause of the *Terrestrial Ecotoxicity Potential* results for the scenario with biosolids use on agricultural soils (Table 6-5). In contrast, the toxicity offsets from avoiding heavy metals in DAP were two orders of magnitude lower than the impacts ascribed to the biosolids. This can be attributed to the high copper (375 mg/kg-ds) and zinc (630 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.

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 in order to avoid the future use of synthetic fertilisers with trace concentration of heavy metals. These fertiliser metal concentrations are expected to increase over time as lower grade phosphate-ore bodies are utilised.

Table 6-5: Detailed breakdown of results for *Terrestrial Ecotoxicity Potential*, for Caboolture Scenario #1 with biosolids to agriculture.

breakdown of total = 3.6 t 1,4-DCB-e/y		
biosolids to soil	avoided fertiliser use	other
85%	0%	15%
organics <1%	Zn <-1%	
Cu 60%	Cd <-1%	
Zn 15%	Hg <-1%	
Hg 4%		
Se 3%		

This exposes the constraints to using the available LCA *Terrestrial Ecotoxicity* models, which are recognised to have major shortcomings in their handling of metals (Pizzol *et al.* 2011). Another barrier is the low coverage, in the available LCA toxicity models, of organic pollutants relevant to wastewater systems management (Lane *et al.* 2012a). This gap needs to be addressed before the relative toxicity contributions of organic and metal pollutants can be meaningfully assessed.

From the available evidence, it appears 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. However, in drawing this conclusion, there are some important developments that the urban water industry should be aware of.

Implications and Opportunities

External forces may have the opposite effect, encouraging the use of LCA for analysis of biosolids use on agricultural lands. The Australian agricultural industry is currently collecting data that will support standardised LCA across a wide variety of Australian crop types and locations (Eady 2012). A key driver for this project is the increasing use of LCA in other countries, to justify food purchasing decisions in markets that currently import substantial quantities from Australian producers. *Terrestrial Ecotoxicity Potential* is one of the impact categories that will be included in this standardised LCA framework, primarily because of the desire to account for the implications of choosing different pesticide management strategies.

Using conventional LCA toxicity models, to assess different crop management practices, could introduce a bias against the choice of biosolids as a crop growth supplement. Such an outcome would conflict with the water industry perception that a shift to agricultural use of biosolids is a positive development in the pursuit of environmental sustainability. Taken to the extreme, it could even reduce the opportunities for agricultural use of biosolids, and increase the need for water utilities to adopt more costly alternatives for biosolids disposal.

One recent development may help to alleviate the mismatch between the LCA models and the local Australian science perspective. A current European research project is developing a globally applicable terrestrial ecotoxicity model that accounts for spatial variability in soil properties, and the effect of this on metals speciation (Owsianiak *et al.* 2012). While this represents a possible step forward in analytical capability, questions remain as to the relevance of this approach for the Australian context. Firstly, because previous studies have questioned whether speciation modelling can adequately predict toxicity effects in many Australian conditions (Menzies 2011). Secondly, the *improved* LCA modelling approach would need to address the issue that marginal additions of some metals (particularly copper and zinc) could have little, or even a positive, effect on the health of soil systems that are otherwise deficient in certain trace metals.

6.3.2. Eutrophication

The next most substantial increase resulting from the shift to agricultural biosolids use is in the *Eutrophication Potential* results (Table 6-3 and Table 6-4). This is a consequence of the European based assumptions that we used for nutrient fluxes to waterways from land application of organic matter (biosolids) and fertilisers. It is likely that these are overestimates of the nutrient fluxes that could be expected in many Australian areas. Leaching fluxes, in particular, are generally considered to be much higher in Europe than in Australia. Nonetheless, the available Australian research has highlighted the potential risk of nitrate leaching from biosolids application under certain conditions, even at 1 NLBAR application rates (Barry *et al.* 2006; Pu *et al.* 2008).

The potential for nutrient leaching and/or surface runoff will vary greatly depending on soil type, climate and farm management practices. Locally specific assumptions would be required for a detailed case study looking at biosolids management options in any particular region. Unfortunately, there is little published material available to guide such an assessment in a standardised manner. Locally specific soil-nutrient modelling might therefore be required to quantitatively predict the net impact of agricultural biosolids use on nutrient fluxes to waterways.

Implications and Opportunities

The trend towards standardised use of LCA to inform Australian agricultural management (see above) will increase the need for such quantitative estimates to be generated. Given the nationwide focus on waterway quality issues, it seems reasonable to expect that nutrient management would be one of the highest profile foci of agricultural LCA.

There remains a major gap in how this could be implemented in practice in Australia. We do not expect that the approach used in this study would be adequate. The large *Eutrophication Potential* result shown in

Table 6-4 illustrates the problems in using the default LCA approach to estimation of nutrient flux from fertiliser applications.

There might, however, be an opportunity to bridge this gap, through interaction with the AusAgLCI project being led by CSIRO (Grant *et al.* 2012). That project is investigating whether the spatially explicit nutrient modelling approach developed for the National Land and Water Resources Audit (Raupach *et al.* 2001) can be modified to give meaningful estimates of nutrient fluxes in a manner suitable for use in LCA studies. If suitable, then extension of this modelling framework to cater for organic fertiliser products would be a beneficial outcome.

6.3.3. Resource Depletion

The agricultural reuse scenario did not deliver any substantial change to the *Minerals Depletion* results (Table 6-3 and

Table 6-4), despite the avoided phosphate rock mining that would otherwise have been required to provide conventional phosphorus fertilisers. This is because the default *Minerals Depletion* model used in this study does not include impact factors for mineral phosphate deposits. Our results therefore do not match the frequently touted ‘sustainability benefit’ attributed to agricultural biosolids use and its capacity to offset phosphate rock mining.

The LCA approach provides an ideal framework for measuring the minerals usage implications over the full life cycle of products and services. In theory, it should also provide a useful framework for comparing the relative significance of depleting different types of mineral deposits. While a number of models have been developed to perform this task, they use a wide variety of methodological approaches. Unfortunately, the LCA research community has not reached a consensus on the best form of metric for this type of analysis.

Furthermore, phosphate resources are left out of most LCA *Minerals Depletion* impact models, as was the case with the ReCiPe model used in this study. To address this gap, we developed an interim impact factor for phosphate rock for use in this analysis. Doing so changes the *Minerals Depletion* results substantially (Table 6-6). The offsets achieved by reducing synthetic phosphate fertiliser use are sufficient to more than compensate for the *Minerals Depletion* burden associated with construction materials, chemicals and services use across the whole wastewater treatment sector.

Table 6-6: *Minerals Depletion* (t Fe-eq/y) results using the ReCiPe model, with and without the inclusion of an interim impact factor for phosphate rock mining.

	Default	With interim P impact factor
total (WW)	117	-27
biosolids disposal	-0.2	-144
sewerage + WWT	6.3	6.4
WW recycling	0	0
infrastructure construction	110	111

However, the benefits attributed to the phosphorus recovery are of the same order as those impacts (downsides) attributed to other aspects of the wastewater system (Table 6-6). Taking this result at face value, it could be inferred that similar ‘sustainability’ benefits might be achieved by reducing chemicals use for wastewater treatment, by choosing different materials for the construction of network or treatment plant infrastructure, or by designing infrastructure systems that are less materially intensive.

The reason for the low significance ascribed to phosphorus recovery is that our interim impact factor for phosphate rock is less than the median impact factor of the ReCiPe *Minerals Depletion* model. The variability in impact factors (~5 orders of magnitude) of this ReCiPe model is typical of other *Minerals Depletion* impact models used in LCA. However, for the three other models that do account for phosphate resources - CML (Oers *et al.* 2002), EDIP (Hauschild *et al.* 2005) and EPS (Bengt 1999) - the phosphate rock impact factor is at least two orders of magnitude lower than the median value for that specific model. In other words, those models treat phosphate resources as much less significant (relative to other minerals) than does the amended ReCiPe model. If used in our analysis, those alternate *Minerals Depletion* metrics would make the phosphorus recovery seem *even less*, rather than *more*, important.

Implications and Opportunities

In an attempt to move beyond the unresolved methodological debates about minerals depletion modelling in LCA, the EU-funded LC-Impact project is developing a new approach in collaboration with a wide range of minerals industry stakeholders (Vieira *et al.* 2011). While the phosphorus-related gaps are recognised by the LCA research community, phosphorus is not being treated as a high priority substance for that project. In part, this reflects a paucity of publicly available data for phosphate rock deposits.

Application of LCA to biosolids disposal analysis would be greatly enhanced if this project were to provide high quality modelling of the implications of phosphate rock depletion, using the best available resource data.

It would also be beneficial to identify and articulate which components of the phosphorus challenge are addressed by this new methodology for *Minerals Depletion* assessment. Phosphorus has a somewhat unique role amongst the broader set of minerals used by human society, because of its fundamental role in food production. As a result, a wide range of socio-environmental concerns have been associated with the prospect of dwindling phosphate rock supplies (Cordell *et al.* 2009).

6.4. GHG Profile of Biosolids Reuse

GHG analysis of agricultural biosolids reuse has traditionally focussed on a comparison of the biosolids transport requirements vs. the avoided manufacture and supply of synthetic fertilisers. A breakdown of the GHG profile for the agricultural disposal scenario (Table 6-7) shows that these benefits (-127 kg-CO_2e/t-ds) are cancelled out by transport over a distance of 150 km (+146 kg-CO_2e/t-ds).

Fugitive emissions from the nitrogen and carbon cycles in agricultural soil are potentially just as significant to the GHG considerations. Using our assumptions, the net increase in N_2O generation (+138 kg-CO_2/t-ds) is of a similar magnitude to the greenhouse significance of the transport. This is largely the difference between the estimated flux from the biosolids nitrogen (279 kg-CO_2/t-ds) and the estimated offset from avoiding the use of synthetic fertiliser (-139 kg-CO_2/t-ds). Both these estimates are extremely uncertain, and likely to be very dependent on crop type, site location, and prevailing management practices. Furthermore, it is questionable whether it is realistic to assume that the same fraction of applied nitrogen will be lost as N_2O, regardless of whether the source is biosolids or synthetic fertiliser. The typical fertilisation approach is to apply nitrogen in ammonium form, whereas biosolids predominantly contains organic nitrogen.

The results also indicate that soil carbon balances will be critical to the overall GHG intensity of agricultural biosolids use. Even with our seemingly low estimate for CH_4 generation from the aerobic soil environment, this flux (70 kg-CO_2e/t-ds) made a small but notable contribution to the overall GHG profile. So too did the soil mineralisation of non-biogenic carbon in the biosolids (81 kg-CO_2e/t-ds). However, the potential for soil carbon sequestration might be far more significant. In the absence of more detailed, locally specific data, we assume that our sequestration factor (24% of carbon applied) represents an upper bound of what might be realistic to expect. Using this assumption, the carbon storage reduced the GHG profile by 257 kg-CO_2e/t-ds, which is the second largest GHG flux associated with the biosolids disposal life-cycle.

Table 6-7: Greenhouse gas balance for agricultural biosolids reuse.

		biosolids use	avoided fertiliser use	avoided crop production
field emissions	CH_4	70 (19%)		
field emissions	N_2O	279 (76%)	-139 (-38%)	-2 (0%)
field emissions	CO_2 (non-biogenic)	81 (22%)		
field emissions	C sequestered (biogenic)	-257 (-70%)		
transport of biosolids		146 (40%)		
other (incl. production)		50 (14%)	-127 (-34%)	-1 (0%)
total		**368 (100%)**	**-266 (-72%)**	**-3 (-1%)**
total		**100 kg CO_2e / t-ds (+27%)**		

The GHG benefits accruing from the crop offsets are notable, but less important than other aspects of the system life-cycle. A review of a wide range of crop production inventories suggests that the GHG offset calculated here might be at the upper end of the possible spectrum, hence our crop offset result might represent an upper bound estimate. This suggests that yield increases will be less important to the overall GHG implications than will the ability to offset synthetic fertiliser use.

To a degree, this latter finding contrasts with the direction of the available biosolids research in Australia. Existing research has focussed more on the demonstration of yield increases than on assessing long term changes in fertilisation practice, in recognition that yield improvements can deliver substantial financial benefits to those farmers making use of the biosolids. In one case, this was estimated to be equivalent to the cost savings associated with a short term reduction in fertiliser application (Barry *et al.* 2006). This disparity highlights that there may not be a good correlation between the financial and the environmental benefits associated with biosolids reuse.

6.5. The Difficulty in Estimating Offsets

The majority of assumptions used in this analysis are extremely variable and/or uncertain, making generalised quantitative analysis difficult. In most cases, this uncertainty could be reduced through continued research into the biophysical aspects of biosolids use on agricultural soils.

However, uncertainties associated with estimating fertiliser and crop offsets may be as much a function of socio-economic characteristics of the agricultural system. This introduces an extra level of difficulty for LCA studies, given LCA practitioners and water system planners are typically more knowledgeable in the biophysical aspects of the system under study. Furthermore, the available biosolids research gives little guidance about how to quantitatively incorporate these aspects into a quantitative study on the overall environmental implications of biosolids use on agricultural fields.

Unless more definitive guidance on the estimation of fertiliser and crop offsets is available, it will be difficult to reach robust conclusions on the merits of the agricultural disposal option for biosolids. Varying the assumptions for effective phosphorus fertiliser offsets (Table 6-8), and crop yield benefits (Table 6-9), highlighted the sensitivity of the LCA results to estimates for these parameters. In both cases, the variation caused notable changes in multiple impact categories, in some cases by ±50% or more.

Table 6-8: Change in results (per kg-ds) with different assumptions on the quantity of synthetic phosphorus application that would be avoided over the long term (default=75% of bioavailable P applied; 'low'=10%; 'high'=100%).

	Eutrophic'n Potential (g PO4-e)	Freshwater Ecotox (g 1,4-DCB-e)	Terrestrial Ecotox (g 1,4-DCB-e)	Global Warming (g CO2-e)	Ozone Depletion (mg CFC11-e)	Fossil Fuel Depletion (g oil-e)	Minerals Depl'n (+P) (g Fe-e)	Human Toxicity (g 1,4-DCB-e)	Particulates formation (g PM10-e)
high fertiliser offset	3.6 (-17%)	0.29 (-8%)	3.4 (0%)	89 (-11%)	8.1 (-1%)	-2 (-186%)	-212 (+33%)	9.2 (-6%)	3.2 (-2%)
default assumption	4.3	0.31	3.4	100	8.2	2	-159	9.8	3.3
low fertiliser offset	6.2 (+43%)	0.38 (+22%)	3.4 (+1%)	128 (+29%)	8.3 (+2%)	12 (+484%)	-22 (-86%)	11 (+15%)	3.5 (+6%)

Table 6-9: Change in results (per kg-ds) with different assumptions on the increase in sorghum yield as a result of biosolids application (default=330 kg-grain/ha; 'low'=no increase; 'high'=660 kg-grain/ha).

	Eutrophic'n Potential (g PO4-e)	Freshwater Ecotox (g 1,4-DCB-e)	Terrestrial Ecotox (g 1,4-DCB-e)	Global Warming (g CO2-e)	Ozone Depletion (mg CFC11-e)	Fossil Fuel Depletion (g oil-e)	Minerals Depl'n (+P) (g Fe-e)	Human Toxicity (g 1,4-DCB-e)	Particulates formation (g PM10-e)
high crop offset	4.3 (-1%)	0.31 (-1%)	3.4 (0%)	95 (-5%)	8 (-2%)	1 (-43%)	-160 (+0%)	9.8 (-1%)	3.2 (-1%)
default assumption	4.3	0.31	3.4	100	8.2	2	-159	9.8	3.3
low crop offset	4.4 (+1%)	0.32 (+1%)	3.4 (+0%)	104 (+5%)	8.3 (+2%)	3 (+43%)	-159 (0%)	9.9 (+1%)	3.3 (+1%)

6.6. Conclusions – LCA and Biosolids Management

The Australian agricultural industry is embracing the LCA concept, which could count against the use of biosolids as a cropping supplement.

The Australian water industry is actively promoting the concept of agricultural biosolids reuse, in order to reduce disposal costs and reduce the strategic risks associated with a dependence on landfill disposal. The industry has invested substantial resources into characterising the risks (environmental and human health) associated with agricultural biosolids application, and using this to inform public debate on the topic.

In parallel, the Australian agricultural industry has begun to embrace the LCA concept, and is currently funding a project that will produce datasets to support standardised LCA across a broad range of agri-systems. The driver for this is the industry's reliance on agricultural exports, and the increasing use of LCA to justify purchasing decisions in overseas markets. It seems likely that LCA will eventually be used, by the agricultural industry and others, to assess whether the use of biosolids will reduce or increase the life-cycle environmental burden of farming.

In theory, the LCA framework is well suited to comparative analysis of different biosolids disposal options, because of its capacity to include a broad diversity of environmental issues. In practice, the current LCA methodologies are not capable of providing robust analysis of biosolids application in three critical aspects. Two of these, relating to biosolids contaminants (*Terrestrial Ecotoxicity*) and the potential for nutrient losses to waterways (*Eutrophication*), will be priority areas for the standardised LCA tool to be adopted by the agricultural industry. The third (*Minerals Depletion*) relates to the displacement of synthetic fertiliser use, and the effect this can have on reducing depletion of phosphate rock reserves.

The analysis provided in this report indicates that the available LCA impact models will tend to make biosolids reuse look substantially less favourable than is indicated by current scientific wisdom. The use of LCA to support agricultural industry planning could, therefore, unfairly bias against the direct application of biosolids to farmlands.

Opportunities exist to leverage off other research projects, and reduce the gap between LCA practice and the available science.

A number of unrelated research projects may provide an opportunity to reduce the LCA-science disparities identified in this report. International and Australian researchers are already working to make general improvements to LCA modelling of metals toxicity, nutrient losses to waterways, and the significance of minerals depletion (or recovery). While this is a welcome development, the current scope of those research projects will not deliver the full set of changes required to support better biosolids analysis.

Opportunities may exist for the Australian urban water industry to further improve the application of LCA to biosolids analysis, by leveraging off that existing research funding. This might alleviate any risk that LCA-based decision making by the agricultural industry will eventually constrain the biosolids management options available to the urban water industry.

Further research is required to inform quantitative, predictive estimates for a number of key operational issues.

Practical application of LCA requires the use of simplified, quantitative estimates for a diverse range of issues, in a form that is not always consistent with the way in which field data is collected.

In terms of GHG analysis, there is a particular need to develop frameworks for estimating the implications of fugitive gas emissions. NGERS based accounting protocols are not adequate to identify and quantify the important GHG fluxes of concern to biosolids analysis. These frameworks should address the significant potential for spatial variability in net fluxes of CO_2, CH_4 and/or N_2O relating to biosolids disposal.

Increases in soil carbon stocks have been shown to occur under Australian conditions, and our analysis using international benchmarks for long term carbon sequestration identified that this could be one of the largest fluxes of relevance to biosolids reuse. Ongoing research efforts must therefore address the question of how much long term sequestration is likely under the variety of soil-climate conditions that apply across Australia. Water utilities should also recognise that some portion of this carbon is likely to be of fossil origin, and therefore not qualify as a net sink of atmospheric CO_2.

Realistic estimates of N_2O generation from biosolids amended fields, and from fields using synthetic fertilisers, will be equally important. To enable this, the industry will need guidance both on the level of emissions, but also on how to predict realistic rates of synthetic fertiliser displacement.

Assumptions on the changes to fertiliser management will also be crucial to estimating net fluxes of nitrogen and phosphorus to water bodies, the level of phosphate rock displacement, and other key environmental outcomes. Field trial studies should be designed not just to quantify short term changes, but also to provide information that can guide assumptions on the longer term implications of biosolids reuse.

The link between biosolids use and increased crop yields is one area that has been given a high priority in current research efforts. This has direct relevance for LCA studies, which should be designed to account for any potential changes to production in other parts of the agricultural system. Further consideration is required on how best to do this in a meaningful way. It will also require far better LCA data for Australian cropping systems than is currently available. However, our analysis suggests there may only be relatively minor GHG benefits achieved by displacing production in marginal cropping areas. The environmental implications of crop yield increases may not be as significant as the direct economic benefits for farmers making use of biosolids as a crop supplement.

7. CONCLUSIONS

This report reviews opportunities for the Life Cycle Assessment (LCA) methodology to inform analysis of conventional urban wastewater treatment systems. Information is presented across five distinct chapters, each focussed on a specific aspect of this investigation.

This section synthesises the findings across the different sections of this report, presenting them in light of the three specific objectives for the study.

Objective 1. Identify key information that LCA can generate to support environmental analysis of wastewater treatment systems.

The case-study applications in this report (Chapters 3, 5 and 6) demonstrate that LCA could play a useful role in supporting environmental analysis for wastewater systems planning. LCA enables consideration of a broad range of environmental issues, beyond just those associated with water use (or reuse) and nutrient discharge. The available evidence suggests that greenhouse gas (GHG) accounting will not be an adequate proxy for the other life-cycle environmental implications of relevance to wastewater system operations. Incorporating additional metrics, based on LCA impact models (see Chapter 4), could therefore help to broaden the horizons of the wastewater industry.

The issue of ozone layer depletion is one that could conceivably come onto the radar of urban water planners in the future (Chapter 3). Anthropogenic Nitrous Oxide (N_2O) emissions will be the greatest source of ozone layer depletion moving into the future, although it remains to be seen whether there will be some form of coordinated international policy response to this issue. Our analysis suggests that, should this happen, this will be an issue of great relevance to wastewater systems assessment. Furthermore, there will be situations where the use of GHG accounting will not encourage minimisation of N_2O emissions. For the sake of minimising strategic risk, it would seem prudent for the industry to avoid (where possible) future increases in N_2O emissions. The research challenges in doing so are discussed below.

Two prominent trends in the Australian wastewater industry are: (a) the ongoing push for increased levels of sewage nutrient removal; and (b) the growing interest in using biosolids as a nutrient source for farming. Life cycle based analysis could provide a valuable addition to consideration on these topics. However, there are important gaps in the available knowledge base, that will constrain the industry from understanding the true environmental implications of both these trends.

In the case of treatment plant nutrient removal, we demonstrate a framework to quantify and interpret the life-cycle environmental tradeoffs associated with reducing nutrient concentrations in treated wastewater streams (Chapter 5). While the case study used in this analysis suggests a strong overall benefit, other studies have raised concerns that advanced nutrient removal could be responsible for a net environmental disbenefit by creating environmental burdens elsewhere. More systematic analysis of different wastewater system configurations would be required to further this debate.

In the case of agricultural biosolids reuse, LCA provides metrics to quantitatively assess all the key environmental issues (Chapter 6). Unfortunately, the results relating to a number of high profile issues (nutrient losses, phosphate recovery, soil toxicity hazard) do not match the current state of scientific thinking, at least in the Australian context. This could potentially create some problems for water utilities in the future, as the Australian agricultural industry is embracing the use of LCA to inform its own management choices. In that context, the shortcomings identified in this report could be expected to bias against the use of biosolids as an agricultural supplement. Improving the impact models available for use in Australian LCA might, therefore ,help alleviate some strategic risk for the urban water industry.

Objective 2. Identify key data requirements for LCA to be implemented, and any gaps in the available data and methodology that would constrain such analysis.

Objective 3. Identify best-available information/assumptions that can be used for quantitative, broad spectrum environmental analysis undertaken by the wastewater sector.

NGERS greenhouse gas accounting protocols will not be sufficient for robust estimation of fugitive gas (N_2O, CH_4, CO_2) emissions from wastewater systems (Chapter 2, 5 & 6). More science based estimates will be required to better understand the possible implications and risks to water utilities. Furthermore, while the recent escalation of research effort has greatly improved the industry's capacity to understand the scale of *overall* fugitive emissions from wastewater systems, there remain some important barriers to use of this information for detailed *options analysis*. This concern is equally relevant for emissions directly from wastewater systems infrastructure, and for those receiving environments affected by the disposal of wastewater or biosolids. Future research should move beyond the characterisation of gross emissions, and provide information that can support simplified, quantitative assessment of different management options. In the meantime, this report provides a review of the best available scientific information on fugitive sources across all aspects of wastewater systems operation.

Scope 3 greenhouse gas footprints might also be an important consideration, when chemicals intensive processing steps are adopted (Chapter 5). Our results suggest that chemicals use for nutrient removal could incur substantial greenhouse gas burdens, although the quality of best practice databases for Australian chemicals manufacturing is uncertain. Research elsewhere has shown that, with the prospect of future changes in electricity and carbon pricing, chemicals supply could represent a significant point of supply-chain price risk. Improved knowledge on chemicals manufacturing operations could therefore be beneficial for the water industry.

The assessment of biosolids disposal options will be particularly sensitive to uncertain estimates for a number of key issues associated with agricultural reuse (Chapter 6). There is a particular need to develop frameworks for estimating fugitive gas emission rates in a way that can adequately represent the significant potential for spatial variability in net fluxes of N_2O, CH_4 and/or CO_2. The same applies for quantifying the long term carbon sequestration potential of biosolids reuse, across the variety of soil-climate conditions that exist across Australia.

Assumptions regarding long term changes in fertiliser management will be crucial to estimating the impacts of agricultural biosolids reuse – particularly for quantifying net N_2O emissions, nutrient fluxes to water bodies, and phosphate rock displacement. For non-experts in agricultural science, there is a notable lack of research available that can support simplified, but realistic, quantitative estimates of fertiliser offsets. The case study analysis provided in this report highlights the range of detailed issues that require consideration, providing one possible set of assumptions for use in future studies. A lack of information will also hamper analysis of the life-cycle implications associated with crop yield increases that result from biosolids application. However, our case study results suggest the life-cycle greenhouse gas benefits might be relatively minor. More extensive research into this topic would be required to fully understand the broader life-cycle implications of such an outcome.

GLOSSARY

General

AWTP	advanced wastewater treatment plant
BAC	biologically activated carbon
GHG	greenhouse gas(es)
IPR	indirect potable reuse
LCA	Life Cycle Assessment
LCI	life cycle inventory
LCIA	life cycle impact assessment
MBRC	Moreton Bay Regional Council
MF	micro-filtration
RO	reverse osmosis
SEQ	South East Queensland
STP	sewage treatment plant
TEQ	toxicity equivalent concentration
TWCMP	Total Water Cycle Management Planning
UF	ultra-filtration
WSI	Water-stress Index

LCA Impact Categories

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

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