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GREENHOUSE GAS INVENTORIES FROM WWTPs - THE TRADE-OFF WITH NUTRIENT REMOVAL

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ABSTRACT

Operating data was collected from a number of wastewater treatment plants (WWTPs) in South-East Queensland and used to calculate greenhouse gas emissions from first principles using an inventory-type approach. Emission factors were based as far as possible on relevant data sourced from either the literature or databases used in Life Cycle Assessment (LCA) models. The results were compared with those from a desktop simulation approach for a range of WWTP configurations that formed the basis for an LCA study of the trade-offs between nutrient removal and total environmental burden. The results from the actual operating plants compared well in general terms with those from the desktop study, although some differences in points of detail were highlighted. In either case, significant uncertainties in GHG emission estimates were apparent in respect of so-called "fugitive" emissions of nitrous oxide and methane from WWTP operations since both of these gases are major greenhouse contributors. Additional research will be required in this area in order to improve the accuracy of emissions reporting from the wastewater sector. Notwithstanding these uncertainties, the data suggested that imported electrical power and biosolids treatment/ disposal operations are the largest potential sources of greenhouse gas. Opportunities for significant reductions (approx. 20 to 40%) in total GHG emissions exist with the inclusion primary sedimentation, anaerobic digestion and power generation from biogas in the WWTP flow sheet. Addition of chemicals (with embodied GHG emissions) to supplement nutrient removal does not cancel out this reduction. However, from a global perspective, the LCA study has shown that GHG emissions represent only a minor (indicatively <1%) of the normalised total environmental impacts from WWTP operations. Eutrophication and potential human health or ecotoxicity impacts associated with the disposal of biosolids (particularly due to the metals content) are the dominant impacts. This raises the question of how policy directives and environmental regulations from government can best serve the complex (and potentially competing aims) of minimising local environmental impacts whilst also improving sustainability on the widest possible front.

KEYWORDS

Greenhouse gas emissions, nitrous oxide, methane fugitive, operating data, wastewater treatment plants, life cycle assessment, impacts, total environmental burden, eutrophication, ecotoxicity

INTRODUCTION

Driven by water quality directives aimed at protecting local aquatic environments from the effects of eutrophication arising from increasing pollution loads of human origin, biological nutrient removal (BNR) is being increasingly applied at wastewater treatment plants (WWTPs) throughout the developed world. When considering the spectrum of wastewater treatment technologies from simplest to most complex, the overall trend is toward increasing energy consumption per unit of wastewater treated. Usually this energy comes from imported electrical power. For example, the simplest lagoon system operated under gravity flow may require little or no electrical power. Preliminary and primary treatment only requires only limited power input (typically <2W/ person equivalent[1]), depending on the method of sludge treatment and disposal. More "advanced" plants consume more power, typically around 4 to 8 W/EP (35 to 70 kWh/ annum per EP) for conventional or BNR activated sludge plants and upwards of 8 W/EP for membrane bioreactors (Hartley, 2007). Hence there is a trade-off between treated effluent quality and power consumption.

Worldwide, most electrical power is generated from fossil fuels and power stations are large contributors to greenhouse gas (GHG) emissions. Hence, simplistically, the effluent quality/power trade-off in wastewater treatment is also a trade-off in terms of greenhouse gas emissions. Currently in Australia, there is growing national political debate over the need for GHG abatement, driven by new government policy directives (NGERS, 2007a; Wong, 2008) and growing public awareness that global climate change is affecting local environments (e.g. record droughts in parts of the country). Public perception of the looming global energy crisis is real but not always well understood (Dukes, 2003). This poses challenges to decision makers or policy regulators since large bodies of fact need to be interpreted and judgement might be blurred by emotions.

Life Cycle Assessment (LCA) is a more comprehensive tool for quantitatively comparing the environmental costs /benefits of different technologies, including wastewater treatment (Foley *et al*, 2007; Gallego *et al.*, 2008). Using so-called "mid point" assessment methodologies, potential life cycle impacts are quantified for a range of environmental categories (e.g. carcinogens, non-carcinogens, ecotoxicity, eutrophication, acidification, global warming and non-renewable energy etc.), in terms of a reference compound (e.g. carbon dioxide equivalents for global warming, phosphate equivalents for eutrophication). These impacts can be further interpreted through an additional so-called "Damage" modelling, which determines the "end-point" impacts on a limited number of core protection areas (e.g. human health, ecosystem quality, resource availability).

However, LCA is computationally intensive and requires detailed inventory information for each technology or application considered. In an effort to build such an inventory toward using LCA, we have found that the collection of data to be a useful starting point, allowing comparisons between different wastewater treatment plants in terms of energy consumption and greenhouse gas emissions. Given the increasing attention to climate change by governments, such data is becoming more important for reporting purposes (e.g. NGERS, 2007a,b). Our experience in Australia has been that collecting and interpreting the relevant data has focussed attention on weaknesses in current guideline methods used by government for estimating greenhouse gas emissions from wastewater treatment processes (Foley and Lant, 2008).

The aim of this paper is to present real operating data from examples of BNR WWTPs in Australia that span a range of sizes. We compare this data with a desktop study that illustrates the trade-off between nutrient removal and GHG emissions or aggregated life cycle impacts. We draw attention to recent studies in Australia aimed at improving GHG accounting methods for WWTPs. Furthermore, we highlight the importance of "gaps" in our knowledge surrounding likely 'fugitive' emissions of nitrous oxide and methane dissolved in the collected raw sewage.

[1] Person equivalent (EP) is expressed in terms of flow (e.g. ~200 L/EP.d) or load (e.g. ~125 g COD/EP.d)

METHODOLOGY

Foley and Lant (2008) compared different methodologies for calculating greenhouse gas emissions from a major Australian WWTP (approx. 1.1 million EP). They concluded that the guidelines published in the federal government's former Australian Greenhouse Office (AGO)[2] workbook, and similar methodologies, were inadequate. The methods provided incomplete guidance and may under-estimate emissions due to misapplication of factors for sludge treatment. These methods also omitted likely fugitive emissions in the form of nitrous oxide (N_2O) and methane (CH_4) from sewers, biosolids treatment and biosolids disposal. For their case study, Foley and Lant (2008) found that potentially these methodologies may predict *negative* CO_2-e GHG emissions (i.e. net uptake in the WWTP) due to inappropriate application of factors in accordance with the published guidelines. Therefore, they recommended the development of a new approach from first principles but identified significant knowledge gaps, particularly in the area of fugitive emissions, which require further research.

In this study a similar approach to that of Foley and Lant (2008), developed at the University of Queensland (Australia), was used. Data was collected from four WWTPs in South-East Queensland. A summary of the key features of these WWTPs is given in Table 1. Inventory data required for calculation of the GHG emissions for these WWTPs is given in Table 2 and a summary of key assumptions is given in Table 3. Using the collected data and assumptions, the method of calculation of GHG emissions was developed from first principles in the functional areas of the plant, namely:

- Raw sewage: methane is likely to be generated in rising mains, mainly due to fermentation and methanogenic bacterial activity in the biofilms attached to the pipelines (Guisasola *et al.*, 2008). This methane will be in dissolved form in the pressurised, full pipes, but will most likely be rapidly released by stripping when the raw sewage is discharged into the WWTP from the pumping mains (Foley and Lant, 2007).
- Screening and grit removal: transport off site to landfill.
- Primary treatment and anaerobic digestion: generating primary sludge that is thickened and anaerobically co-digested with thickened waste activated sludge (where applicable). Anaerobic digestion generates biogas that may be burned in a flare or in engines for on–site generation of electricity. The burned component of methane is greenhouse-neutral (assuming organics are primarily of renewable carbon origin e.g. foods). However, GHG contributions are accounted for in the form of dissolved methane release from the digested sludge, as well as emissions due to inefficiencies of the digestion biogas collection and combustion system, leading to methane losses or nitrous oxide generation during biogas combustion.
- Secondary treatment: Biological oxidation of BOD (biodegradable COD) generates renewable carbon CO_2 that is greenhouse neutral (except where organics of fossil fuel origin are imported e.g. methanol produced from natural gas), but potentially significant fugitive GHG emissions of nitrous oxide from biological nitrification and denitrification (see Table 3).
- Chemical dosing: Transport using fossil fuels produces GHG, the chemical production produces "embodied" GHG emissions that are accounted for at the point of consumption on the WWTP. If organics of fossil fuel origin are used, the oxidised carbon is a source of CO_2 that contributes GHG (but not if from renewable sources e.g. molasses from sugar cane).
- Biosolids: Transport off site using fossil fuels produces GHG, and fugitive emissions of methane and nitrous oxide may occur to a variable (and somewhat uncertain) degree from

[2] Now called the National Greenhouse Accounting method in the Dept. of Climate Change

disposal to either landfill or agricultural use for the WWTPs considered here (see Table 3).

Table 1 - Summary features of the four wastewater treatment plants (WWTPs) surveyed in this study (2006-7 data)

Parameter	**Unit**	**Plant A**	**Plant B**	**Plant C**	**Plant D**
Type of main biological treatment process	-	Extended aeration activated sludge, 3-stage Phoredox-type with Oxidation 'Ditch'	Extended aeration activated sludge, 5-stage Bardenpho-type (compartmentalised)	Primary sedimentation, 5-stage Bardenpho-type (compartmentalised)	Primary sedimentation, 3-stage Phoredox-type (compartmentalised)
Prefermenter to promote biological P removal	-	Yes	Yes	Yes	No
Anaerobic digesters	-	No	No	Yes	Yes
Power from biogas	-	No	No	No	Yes
Effluent filters	-	No	Yes	Yes	No
Effluent disinfection	Type	Chlorine gas	UV	UV	None
Design flow	50%ile, ML/d	7.5	12.0	34.0	170
Current Daily flow	Annual 50%ile, ML/d	7.9	10.4	24.1	123.7
Nominal design capacity	Equivalent persons (EP)	30,000 (to 45,000)	60,000	Approx. 136,500	850,000
Current nominal EP load	EP[3]	47,600	50,000	100,400	581,572
Attributed EP load	EP (load basis)[4]	35,400	54,000	80,800	706,400
Design COD or BOD load	50%ile kg/d (mg O_2/L)	COD 4,125 (550)	BOD 3600 (300)	COD 17,070 (502) 7,113 (209)	COD 120,000 (700)
Current raw COD &/or BOD load	50%ile kg/d (mg O_2/L)	4,424 (560)	COD load[5] 9,755 (938) BOD load 3240 (312)	10,100 (419)	COD load[6] 100,200 (810) BOD load 38,850 (314)
Current raw T(K)N load	50%ile kg/d (mg N /L)	416 (52.7)	634 (61.0)	1,181 (49)	8,024 (64.9)
Current raw TP load	50%ile kg/d (mg P /L)	99 (12.5)	126 (12.1)	176 (7.3)	1,516 (12.3)
Licence (current) effluent TN	50%ile mg N/L	5.0 (1.5)	5.0 (4.95)	3.0 (2.4)	5.0 (4.0)
Licence (current) effluent TP	50%ile mg P/L	2.0 (0.3)	1.0 (0.07)	1.0 (0.4)	10.0 (7.5)
Chemicals dosed for supplementary nutrient removal	Type and reason	Alum (supplementary simultaneous chemical P removal)	Molasses (biological P removal) + Lime (sidestream chemical P removal)	Methanol (N removal) + Alum (supplementary simultaneous chemical P removal)	None

[3] Based on estimates by Council by various methods (flow, tenements or city planning)
[4] Assuming 125 g COD/(EP.d) or 55 g BOD/(EP.d) (i.e. COD/BOD ratio = 2.27)
[5] Uncertain current COD data for Plant B, possibly due to sewer infiltration of saline and coloured groundwater
[6] Uncertain current COD data for Plant D, possibly due to sewer infiltration of saline groundwater

Table 2 - Inventory data for calculation of greenhouse gas emissions from WWTPs in this study. Data are current annual averages (2006-7 data)

Parameter	**Units**	**Plant A**	**Plant B**	**Plant C**	**Plant D**
Total Plant Power[7]	kW	227	297	795	2,343
Aeration Power	kW (approx.)	72	96	328	1,640
UV disinfection power (type)	kW	Nil	73 (medium pressure lamps)	37 (low pressure lamps)	Nil
Power recovered from biogas	kW (% total) average	Nil	Nil	Nil	1,102 (47%)
Other fuels	Type, L/d	Diesel, <1	Petrol, <1	Diesel, <1	Diesel, 263
Imported chemicals					
Organics	Type, L/d	Nil	Molasses,	Methanol,	Nil
Alum	L/d	650	487	775	Nil
Lime	kg/d	175	Nil	2,550	Nil
Polymer	kg/d	9	200	65	438
Chlorine gas	kg/d	44	18	60	Nil
			Nil	Nil	
Nitrogen removal by denitrification[8]	kg/d (% of TN load)	264 63%	358 56%	750 64%	5,925 74%
Screenings & Grit	kg/d wet	272	286	1,295	4,400
Biosolids disposed	kg/wet kg/d dry	12,529 2,130	25,714 3,214	16,693 5,843	118,904 22,592
Biogas produced	kL/d	Nil	Nil	2,900	13,302
Methane in biogas	% v/v	-	-	Assume 65%	67%
Sludge[9] flow via anaerobic digesters	kL/d primary kL/d WAS	Nil	Nil	171 (combined)	407 427
Flow via rising mains[10] to WWTP	% of total (approx.)	100%	80%	100%	100%
Biosolids disposal	Type	Landfill	Agriculture	Agriculture	Minesite rehabilitation/ Landfill

[7] Excluding lift pumps (raw sewage, &/or filter feed/ final effluent)
[8] Calculated by difference: (Influent TN – Effluent TN – Biosolids TN)
[9] Both primary sludge and waste activated sludge (WAS) are thickened pre-anaerobic digestion on Plant D
[10] Remainder via gravity mains

Table 3 - Assumptions for greenhouse gas emission calculations (common to all WWTPs).

Parameter	Assumed value	Units	Reference/ Source	Comments
Emission factor for imported electrical power	1.04	$kgCO_2$-e/kWh	NGA (2008)	Queensland, Full fuel cycle
Emission factor for other fuels	Diesel: 2.3 Petrol: 2.7	$kgCO_2$-e/ L	NGA (2008)	Full fuel cycle
Emission factors for production of chemicals	Molasses: 0.2 Alum: 0.539 Lime: 1.640 Polymer: 1.182 Chlorine gas: 1.124	kgCO2-e/kg kgCO2-e/kg dry kgCO2-e/kg dry kgCO2-e/kg dry kgCO2-e/kg dry	NGA (2008)	For molasses: 50% of listed value for ethanol (NGERS, 2007) Other values from *SimaPro*® v7.1.0 Australian LCA data library (2007).
Fuel efficiency	0.554	L/km	Giannelli et al. (2005)	Heavy diesel truck
Delivery distances	20 to 500	km (approx. for round trip)	WWTP operator	Varies for chemicals (origin) or biosolids & screenings etc. (destination)
Global Warming Potentials (GWP)	CO_2: 1 CH_4: 25 N_2O: 298	$kgCO_2$-e	IPCC (2007)	GWP-100, 100 year horizon
Emission factors for nitrous oxide (fugitive)	Secondary Treatment Off-gas: ≤0.01 - 0.05 Landfill/ Minesite: 0.0082 Agriculture: 0.0159	kgN_2O-N/kg N denitrified kgN_2O-N/kg N as biosolids disposed As above	Various refs.[11] Various refs.[12] As above	Biological nitrification-denitrification in WWTP or soil/ landfill biological processes
Emission factors for methane (fugitive)	Digester/ biogas leaks: 1% Unoxidised CH_4 in combustion: 0.0034 Dissolved CH_4 in digested sludge: 20 Dissolved CH_4 in raw sewage: 5 - 30 Landfill/ Minesite: 0.00283 Agriculture: 0.0041	Percent of biogas produced $kgCH_4/kgCH_4$ burned mg/L CH_4, 35degC mg/L CH_4, 25degC (approx.) $kgCH_4$/kg dry biosolids disposed As above	Bridle (2007) Doka (2003); Smith *et al.* (2000) Calculated (without super-saturation) Guisasola *et al.* (2008) Various refs.[13]	Anecdotal Zimmermann et al. (1996), *in* Doka 2003) Henry's Law, assuming gas-liquid equilibrium

[11] Refer to Foley and Lant (2007, 2008): Eleven refs. for municipal wastewater treatment, Range 0.0003 – 0.05 (Median 0.01) kgN_2O-N / kgN influent.
[12] Refer to Foley and Lant (2007, 2008): Three refs. for N_2O from Landfill, Range 0.002 to 0.016 kgN_2O-N / kgN disposed; Eleven refs. for N_2O from Agriculture, Range 0.006 to 0.035 kgN_2O-N / kgN applied.
[13] Refer to Foley and Lant (2007, 2008): Three refs. for CH_4 from Landfill, Range 0.001 to 0.089 $kgCH_4$ / kg dry waste disposed; Three refs. for CH_4 from Agriculture, Range Negligible to 0.0096 $kgCH_4$ / kg dry biosolids applied.

Table 4 - Results of GHG calculations for four scenarios using methodology of this study, given as percentages of Total GHG emissions, except Totals (in tonnes CO2-e-annum).

Scenario	**Sec. Off-Gas N_2O kgN/kgN denitrified**	**Sewage CH_4 mg/L**	**Category**	**Plant A**	**Plant B**	**Plant C**	**Plant D**
1	0.01	5	Power	53%	51%	59%	30%
	(Low)	(Low)	Chemicals & Fuel	5%	4%	7%	2%
			Sec. Off-gas (N_2O)	12%	12%	11%	27%
			An. Dig./Biogas (CH_4)	0%	0%	2%	3%
			Biosolids	21%	25%	12%	23%
			Sewage (CH_4)	9%	7%	9%	15%
			Total	**3,892**	**5,264**	**12,289**	**37,152**
2	**0.05**	5	Power	36%	35%	42%	15%
	(High)	(Low)	Chemicals & Fuel	4%	3%	5%	1%
			Sec. Off-gas (N_2O)	40%	40%	37%	65%
			An. Dig./Biogas (CH_4)	0%	0%	1%	1%
			Biosolids	14%	17%	9%	11%
			Sewage (CH_4)	6%	5%	6%	7%
			Total	**5,696**	**7,711**	**17,419**	**77,660**
3	0.01	**30**	Power	36%	38%	41%	17%
	(Low)	**(High)**	Chemicals & Fuel	4%	3%	5%	1%
			Sec. Off-gas (N_2O)	8%	9%	8%	15%
			An. Dig./Biogas (CH_4)	0%	0%	1%	2%
			Biosolids	14%	19%	9%	13%
			Sewage (CH_4)	38%	32%	37%	52%
			Total	**5,695**	**7,162**	**17,787**	**65,377**
4	**0.05**	**30**	Power	28%	28%	32%	11%
	(High)	**(High)**	Chemicals & Fuel	3%	2%	4%	1%
			Sec. Off-gas (N_2O)	30%	32%	28%	48%
			An. Dig./Biogas (CH_4)	0%	0%	1%	1%
			Biosolids	11%	14%	7%	8%
			Sewage (CH_4)	29%	24%	29%	32%
			Total	**7,498**	**9,609**	**22,917**	**105,885**

Note: Minor percentage fraction for screenings & grit removal not shown (<0.1% of total) but included in Totals.

RESULTS AND DISCUSSION

The results of GHG calculations are presented in Table 4 for four different scenarios of assumed nitrous oxide and methane fugitive emissions. Scenario 1 results are plotted in Figure 1 and Figure 2.

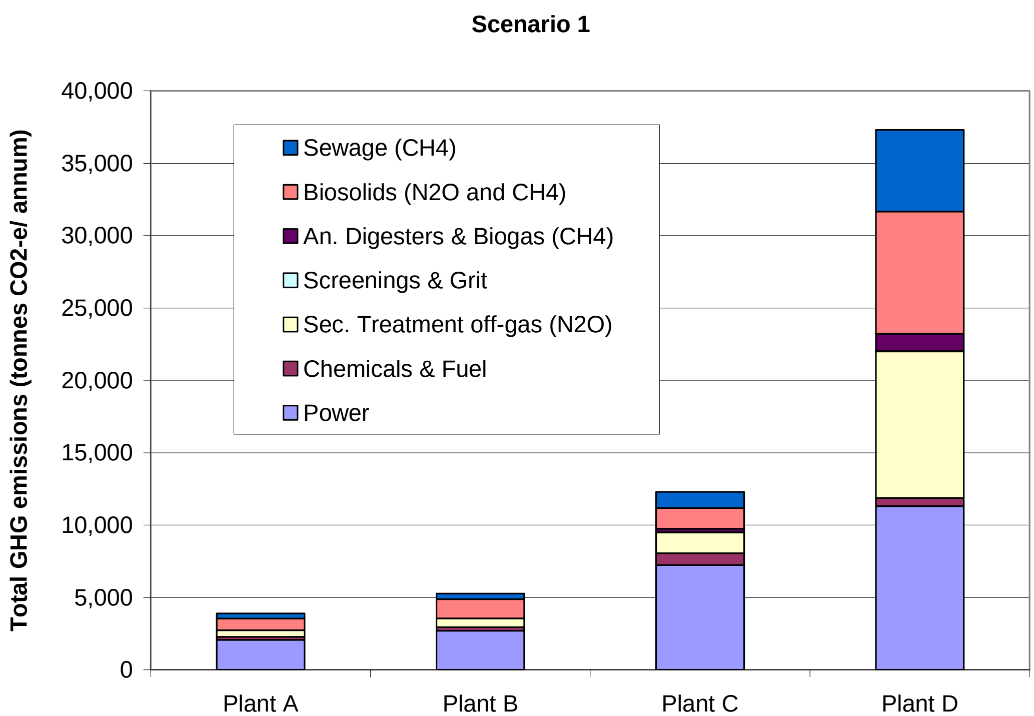

Figure 1 - Summary of Greenhouse Gas emissions calculated using methodology of this study.

The results suggest that GHG emissions are dominated (on a percentage basis) by imported power, followed by fugitive emissions of nitrous oxide (from nitrification-denitrification in the secondary process or biosolids disposal i.e. denitrification in the soil after agricultural use or in a landfill operation). The proportion of total GHG emissions from power is significantly reduced in Plant D (30%), due to on-site power generation using biogas, compared with Plants A to C (~50 to 60%) which either do not have anaerobic digesters to produce biogas or lack power generation equipment. This is also immediately apparent when GHG emissions are compared on a flow or load-specific basis (Figure 2). For example, on a flow-specific basis, Plant D produces 46% less GHG compared to Plant C (which has a comparable process configuration), and 40% less GHG compared to Plants A and B (extended aeration-type plants)[14].

Obviously, anaerobic digestion with electrical power generation from biogas offers an opportunity for significant GHG reduction (by at least roughly one third). In this study, the comparisons may be somewhat skewed by the differences in plant sizes since large plants might give power savings, on a relative basis, due to economies of scale. Nevertheless, from a desktop study for equally sized plants, de Haas and Hartley (2004) found that GHG reductions in the range 23% to 38% should be possible with anaerobic digestion compared to extended aeration. They also pointed out that the embodied GHG contributions due to additional chemical

[14] The GHG emissions from Plant D are 35% lower than a sister plant to Plant A, which is also an extended-aeration type plant with an average daily flow of 6 ML/d (data not given, see error bar in Figure 2).

consumption (methanol and alum) required to achieve comparable nutrient removal (effluent Total N <= 3 mgN/L; Total P <= 2 mgP/L) with anaerobic digestion did not cancel out the benefit in terms of overall GHG emissions.

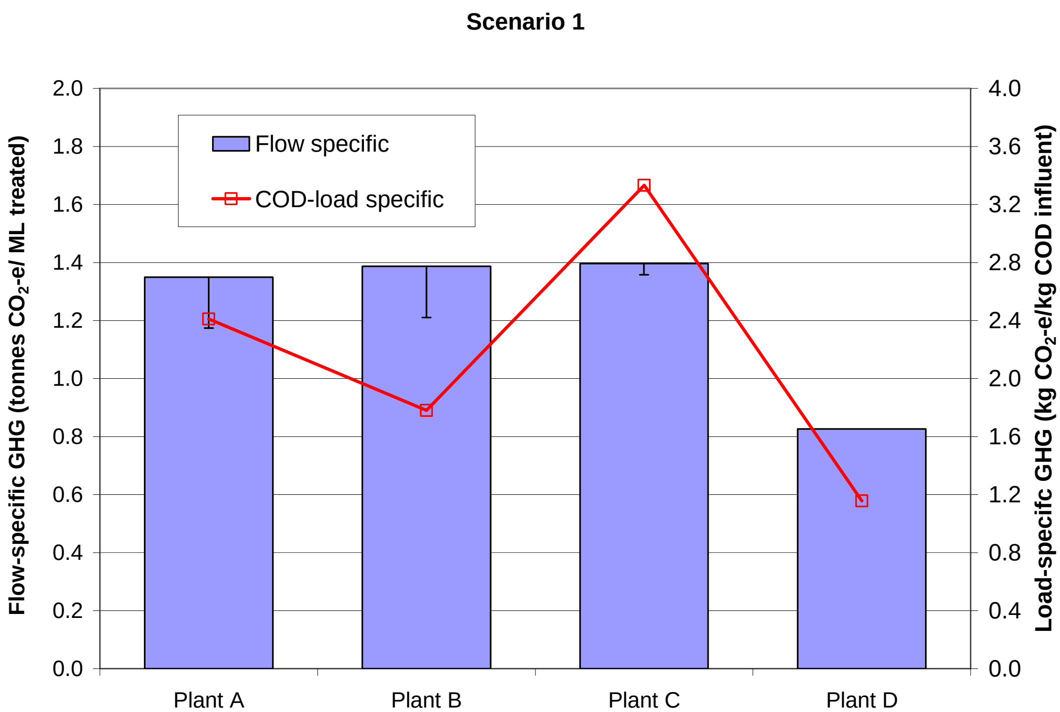

Figure 2 - Comparison of GHG emissions on a flow or COD load-specific basis using methodology of this study. Error bar for Plant A: flow-specific range for similar sister plant (ADWF 6.0 ML/d). Error bars for Plants B & C: excluding average power consumption for UV disinfection. COD loads for Plants B and D estimated from measured BOD load data, assuming COD/BOD ratio = 2.27.

The real operating data collected in this study have confirmed the desktop findings of de Haas and Hartley (2004). For example, Plant C (anaerobic digestion without power recovery from biogas, but with methanol, alum and a little lime dosing for supplementary nutrient removal) produces similar GHG emissions on a flow-specific basis to Plants A and B (Figure 2). Plants A & B are extended aeration plants that achieve comparable nutrient removal through biological means with supplementary chemicals (either alum for P removal or molasses for N &/or P removal). If the power component for UV disinfection in Plants B & C is excluded from the calculations (see error bars in Figure 2), then the emissions are somewhat reduced, especially for Plant B[15], but the emissions from Plant C are still comparable with those of Plants A or B.

The licence for Plant D does not require P removal but the plant has had a number of upgrades to its activated sludge process in order to optimise N removal. These have resulted in Plant D most recently achieving an effluent Total N concentration comparable to the other plants considered here without the need for chemical dosing (Table 1). Overall, for Plant D, the contribution from embodied emissions in chemicals dosed (including polymers for sludge thickening/dewatering and small amounts of fuel used on the plant for maintenance operations e.g. stand-by generators

[15] Due to the relative inefficiency of the older medium-pressure UV lamps on this plant.

and back-up heating of digesters) accounted for 2 to 7% of total GHG emissions, which is relatively little.

Emissions of nitrous oxide from secondary off-gas (activated sludge denitrification process) are uncertain but potentially substantial (Table 4). Taking Scenario 1 (1% N_2O-N formed of nitrogen denitrified in treatment process), the relative contribution to total GHG emissions from secondary off-gas N_2O is higher for Plant D (27%) compared with the other plants (11-12%) due to the reduction in total emissions for Plant D from biogas power recovery (see above). In Scenario 2, assuming 5% N_2O-N formed of nitrogen denitrified in treatment process, the relative contribution to GHG emissions from this source increases to roughly 40-65%, and the total annual emissions increase by nearly 50% (Table 4).

Similarly, emissions of unoxidised methane, which is likely to be stripped to atmosphere from dissolved form in the raw sewage (Guisasola *et al*., 2008), may represent between approximately 5-15% of the total GHG emissions if present at relatively low concentrations (5 mg/L CH_4 in Scenarios 1 & 2, Table 4). However, this might increase significantly if present at higher concentration (e.g. approximately 24-52% of total GHG emissions if present in sewage at 30 mg/L dissolved CH_4 - see Scenarios 3 & 4, Table 4). The sewage dissolved methane alone has the potential to increase total emissions accounted for at the WWTP by approximately 10% to 100%, depending on the amounts present (range 5-30 mg/L) and the reference case (e.g. extent to which N_2O is formed – see above).

The data collected in this study suggests that nitrous oxide and methane arising from biosolids disposal account for similar proportions of total GHG emissions as N_2O from the secondary treatment process (Table 4). The literature (see references in Table 3) shows that there are also large uncertainties in emission of these fugitive gases from biosolids disposal. The range for N_2O appears to be approximately 0.2 to 1.6% of N applied as biosolids for agriculture and ~0.6 to 3.8% on the same basis for landfill. For methane the range appears to be ~0 to 10 g CH_4/ kg dry biosolids applied or agriculture and ~1 to 90 g CH_4/ kg on the same basis for landfill.

Clearly, further research will be required in the area of fugitive emissions particularly of nitrous oxide and methane in order to improve the certainty of GHG accounting from sewage collection and WWTPs.

Despite the limitations arising from knowledge gaps or uncertainty due to fugitive emissions, GHG estimation has commenced in Australia and reporting will become mandatory for many institutions from July 2008 (NGERS, 2007a). Parallel work will occur in research areas to improve the estimation methodology and accuracy. A life-cycle-type approach is a useful analytical tool for comparing different scenarios or processes (e.g. types of WWTP, management options for biosolids disposal or level of nutrient removal). In the lead up to such work, it is useful to have gross conversion factors (or “rules of thumb”) in order to estimate the relative significance of components in a large network (e.g. water supply and treatment, wastewater collection and treatment, and possibly water recycling). Preferably these “rules of thumb” should be benchmarked against actual operating data.

From the data collected in this study, it appears that flow-specific GHG emissions, expressed as tonnes of CO_2-equivalents per megalitre sewage treated, is a useful basis for comparing plants. Figure 2 suggests that despite a 16-fold range in raw sewage flows treated between Plants A[16] to

[16] The range is approximately 20-fold if the sister plant to Plant A-1 is included (all data not given for Plant A-1, which is similar to Plant A, but with ADWF = 6 ML/d and including sand filters for the secondary effluent).

D (or indicatively approximately 30,000 to 700,000 EP), the range in GHG emissions on a flow-specific basis is less than 2-fold, i.e. ~0.8 to 1.4 tonnes CO_2-e/ ML for Scenario 1 fugitive emissions (see above). As noted before, some economies of scale may be expected. A larger data set from this study (not fully reported here) suggests that smaller plants (down to roughly 2000 EP) may produce up to 2.5 tonnes CO_2-e/ ML on the basis of the same assumptions. Nevertheless, despite range of between 250 to 350-fold in flow or EP, the range for the larger data set in GHG emissions on a flow-specific basis is only about 3-fold.

It is interesting to also compare GHG emissions on a load-specific basis. For example, Figure 2 shows emissions expressed as tonnes/ CO2-e/ kg influent COD. This shows a greater variation than on flow-specific basis (range 1.2 to 3.3 tonnes/ CO2-e/ kg influent COD). This is caused by the greater variability in measured 50%ile influent COD concentration (range ~420 to 710 mg/L from data[17] in Table 1), which may be partly due to inherent difficulties associated with accurate sampling of raw sewage from WWTPs. The fact that the GHG emissions appear to be more constant on a flow-specific basis may be partly a reflection of the fact that a significant part of the total plant power (approx. 30 to 55%, excluding lift pump stations) is not aeration-related (e.g. mixing, pumped recycles, lighting, dewatering) and therefore largely independent of COD load. Furthermore, the major aeration component of total plant power (45 to 70%) is dominated in nutrient removal plants of the type considered here, by oxygen demand for nitrification, which is proportional to influent TKN load. Reported influent TKN concentrations (49 to 65 mgN/L) were less variable than COD or BOD. However, for a number of reasons, many plants are less likely to have accurate influent COD, BOD or TKN data than influent flow data. Hence a load-related “rule of thumb” for GHG emissions may in practice be less useful than a simpler one that is flow-related.

COMPARISON WITH DESKTOP LIFE-CYCLE ASSESSMENT STUDY

Foley (2008) has studied WWTPs of various configurations using a desktop modelling and Life Cycle Assessment (LCA) approach. The inventory of data used was based on experience in the design and operation of WWTPs but standardised in terms of inputs and outputs using the BioWin® simulation software. The BioWin model outputs were interpreted in terms of so-called LCA ‘mid-point’ impact categories using *SimaPro®* software (see details below). The period assumed for the LCA model was 15 years, which corresponds to an indicative minimum life typically applied for planning and asset maintenance purposes at WWTPs in Australia.

Characterisation: Impacts from greenhouse gas emissions

Climate change/ global warming (GHG emissions) is one of the mid-point impact categories used in LCA. The outputs from the LCA data of Foley (2008) were compared with the actual plant data and associated GHG estimates for Plants A to D in this study (see above). The results are shown in Figure 3. The actual plant data show reasonably good general agreement with the desktop LCA data (Foley, 2008) but differ on points of detail:

- The LCA data included allowance for the emissions during construction of the plant, including embodied emissions for the construction materials, which were not included in the calculations based on operating data for actual WWTPs. However, the construction-phase contributions typically account for <5% of the total emissions over the life of the

[17] Assuming COD/BOD ratio 2.27 for conversion where only measured influent BOD data was available.

15-year life of the plant considered. This is a relatively minor component and would diminish if longer life cycles were considered.

- The LCA data included allowance for avoided artificial fertilizer use in proportion to the nutrient content (N & P) of biosolids applied in agriculture. Agricultural application was applied for all the plants. For the actual WWTPs considered here, Plants B & C have disposal of biosolids to agriculture, whereas Plants A & D have disposal to landfill or minesite rehabilitation. Avoided fertilizer use was therefore not considered in the emission calculations based on the actual WWTP operating data. The approximate margin for predicted GHG emissions from the desktop LCA data due to avoided fertilizer use was - 2.0 – 2.5% of the total emissions (or about 0.03 tonnes CO_2-e/ML treated).
- The Oxidation Ditch plants (Plant A and a similar sister plant A-1) produce lower 50%ile total N effluent concentrations (~1.5 to 2 mgN/L) than the lowest modelled in the LCA study (3 mgN/L). The LCA model predicted lower GHG emissions (~1 to 1.1 tonnes CO_2-e/ML) compared with ~1.2 to 1.35 tonnes CO_2-e/ML based on actual operating data for these plants. Apart from the differences in approach already mentioned above, this difference may also be due to higher actual total plant power consumption being approximately 25% higher than was actually modelled (i.e. 689 kWh/ML from data in Table 1 and Table 2 vs. approximately 550 kWh/ML modelled). Economies of scale (6 to 8 ML/d actual ADWF compared to 10 ML/d modelled) may also account for some of the differences.
- The compartmentalised Bardenpho-type plant (Plant B) is of very similar size to the desktop study size chosen (ADWF 10 ML/d). The GHG estimates for this plant showed good agreement with the LCA model predictions (1.1 vs. 1.2 tonnes CO_2-e/ML treated). UV disinfection power was excluded for the purpose of this comparison and the remaining differences are likely to be due to be due a combination of the above-mentioned differences in approach, as well as actual vs. model assumptions for power (685 kWh/ML actual vs. approx. 624 kWh/ML treated) and chemical consumption. For example, the actual plant uses aerobic digestion for stabilising waste activated sludge (WAS), which will use more power than the method of WAS stabilisation modelled (sludge lagoons). Furthermore, the actual plant achieves excellent bio-P removal by means of a combination of raw sewage pre-fermentation and dosing molasses (a source of readily biodegradable COD), whereas the model assumed a combination of supplementary methanol dosing for N removal and alum for P removal.
- The LCA study examined a case of primary sedimentation and anaerobic digestion (including power recovery) followed by a "modified Ludzack-Ettinger" (MLE) activated sludge configuration plus supplementary chemical dosing for low TN and TP concentrations (<10 and <5 mg/L respectively). This is closely similar to Plants C and D, although Plant C does not have power recovery. Moreover, the exact plant process configurations for these plants are not MLE (i.e. two-stage, anoxic-aerobic) but rather three or five-stage modified Bardenpho types[18]. In the case of Plant D, the GHG estimates from actual plant data were significantly lower than the model predictions (0.8 vs. approx. 1.25 tonnes CO_2-e/ML) and the actual plant has recently achieved a slightly lower effluent TN (4 mgN/L) than the lowest modelled (5 mgN/L). In the desktop study it was found that, using the BioWin activated sludge simulator, an MLE configuration required relatively large methanol doses to achieve an effluent TN concentration as low as 5 mgN/L. In practice at Plant D, after recent upgrades using the modified Bardenpho

[18] The modified Bardenpho (or Phoredox) configuration was not modelled together with primary sedimentation, anaerobic digestion and biogas recovery. The next nearest configuration modelled was the MLE-type which was therefore used for comparative purposes here.

configuration with careful aeration and recycle control, effluent TN <5 mgN/L have been achieved without chemical supplement[19].

- The main cause of the difference in the GHG estimates for Plant D is likely to be economies of scale since the actual plant is about 12-times larger than that modelled. In the case of Plant C (which is only roughly twice as big as that modelled), the agreement was much better (~1.3 to 1.4 tonnes CO_2-e/ML), except that the actual plant achieves a lower effluent TN than the lowest modelled (2.4 vs. 5 mgN/L). This may be due to a combination of model limitations and differences in actual influent characteristics (e.g. lower actual influent TKN concentration).

None of the plants surveyed in this study included membrane bioreactors (MBR). However, it is worth noting that the desktop LCA study predicted that extended aeration MBR-type plants were predicted to have higher GHG emissions than more traditional activated sludge configurations (Figure 3), due to higher power and chemical consumption to achieve comparable nutrient removal. MBRs do, however, offer benefits of higher water quality that is more suited to water recycling.

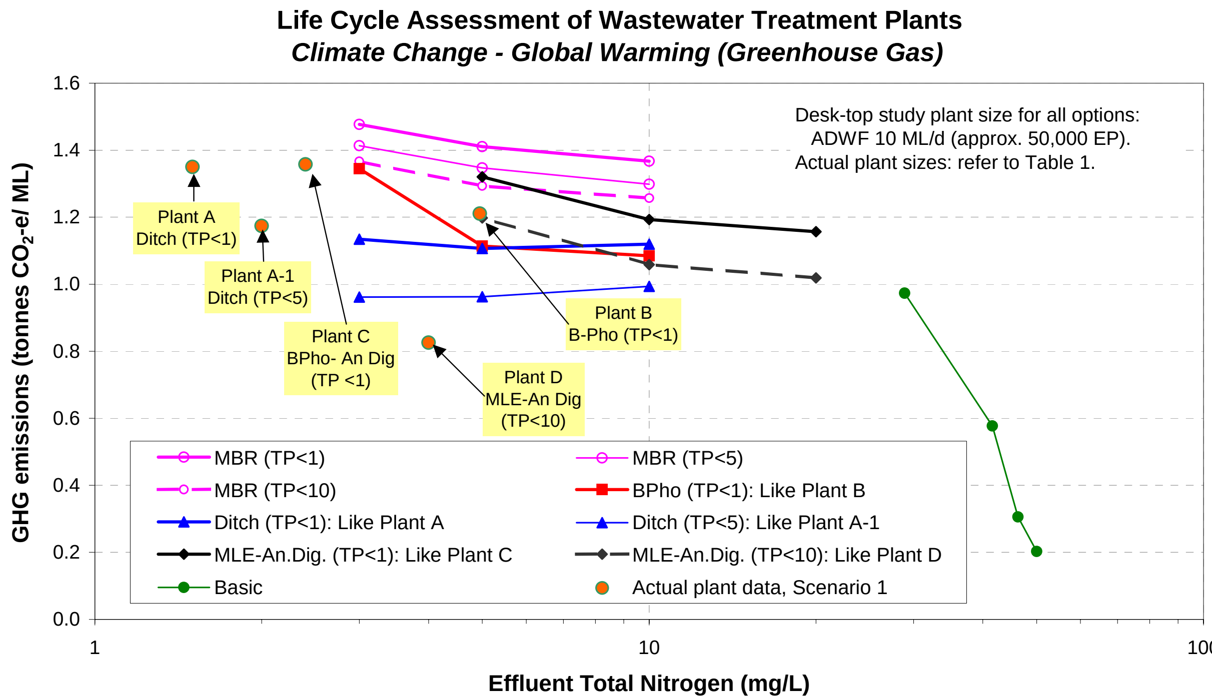

Figure 3 - Comparison of GHG emission calculations from desktop LCA study with actual plant data collected in this study (assuming Scenario 1 fugitive emissions – refer to Table 4)

Normalisation and Total Environmental Burden (TEB)

The desktop study applied the IMPACT 2002+ Life Cycle Impact Assessment (LCIA) (Jolliet et al., 2003) method in the *SimaPro*® software. Using this method, impacts at the so-called "mid-point" are calculated for a range of categories over the chosen life cycle of the plant. The impacts

[19] Prior to the most recent plant upgrade (2007-8), Plant D achieved a median effluent TN of 7.7 mgN/L (2006-7 data), also without chemical supplementation.

are expressed in terms of equivalents of a reference compound or norm for a given category, for example: carbon dioxide (kg CO_2-e) for Global warming; orthophosphate (kg PO_4-e) for Eutrophication; Triethylene Glycol (kg TEG-e) for Terrestrial or Aquatic Ecotoxicity; and arable land area (m^2) for Land Occupation. The complete list of LCIA fifteen categories applied in this method is as follows:

- Carcinogens
- Non-Carcinogens[20]
- Respiratory inorganics
- Ionizing radiation
- Ozone layer depletion
- Respiratory organics
- Aquatic ecotoxicity
- Terrestrial ecotoxicity
- Terrestrial acidification/ nutrification[21]
- Land occupation
- Aquatic acidification
- Aquatic eutrophication
- Global warming
- Non-renewable energy
- Mineral extraction

For any process (e.g. WWTP in this study), the impacts will be greater in some LCIA categories than in others. For example, on the one hand WWTPs are potentially the major point sources of eutrophication for a given population served where most of the nutrients excreted are directed via wastewater to the WWTP. On the other hand, a WWTP is not expected to be significant source of ionising radiation and its land occupation will be small compared to other human activities (homes, roads, buildings etc.). Intuitively, then, if a given WWTP studied is representative of (or similar to) the type of treatment serving a population on a national basis, then normalisation of LCIA data for the WWTP relative to a national database offers a rational approach for interpretation. In this study, the desktop study WWTP LCIA data was normalised relative to the all the recorded environmental impacts caused by a serviced population of 50,000 “average” Australians. Effectively, this expresses the contribution from the WWTP in each environmental impact category as a fraction of the total emissions in that category by the population served. This gives a dimensionless percentage for each impact category, which allows the results to be aggregated under any given weighting scheme.

Commencing with the relative contributions in each of the LCIA categories with respect to the sum of the normalised impacts for the WWTP, it becomes apparent that the WWTP has major impacts in four dominant categories[22], namely:

- Human health effects due to Non-Carcinogens (principally heavy metals in biosolids);

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[20] Non-carcinogens are compounds that impact on human health by potentially causing illnesses other than cancer. Metals (including ‘light’ metals such as aluminium from alum addition during treatment, and heavy metals found in elevated concentrations in wastewater) are a typical example of relevance to WWTPs. The human health impacts are assessed using a combination of fate, exposure and effects models. These include assumptions on metal bioavailability for uptake by plants used in food production after agricultural application of biosolids.
[21] Acidification and nutrification are caused by depositions of organic substances, such as sulphates, nitrates and phosphates. These depositions mainly occur through air and directly into water. The primary effect is the change in nutrient level and acidity in the soil (Jolliet et al, 2003).
[22] Similar conclusions were reached by Gallego *et al.* (2008) in their study of WWTPs of <20,000 EP size in Spain.

- Aquatic ecotoxicity (residual metal concentrations in run-off from biosolids land application and effluents released to receiving waters);
- Terrestrial ecotoxicity (principally effects of metals from biosolids disposal); and
- Aquatic eutrophication (mainly nutrients containing N & P in effluents released to receiving waters).

Figure 4 shows that in percentage terms, relative to the sum of the normalised impacts for two of the WWTP cases[23], these categories account for >90% of the total estimated normalised impacts from the WWTP. The other categories collectively account for <10% of the impacts; in some categories the impacts are negligible (effectively zero). Interestingly, Global Warming (GHG emissions) account for around 0.4 to 0.8% of the total normalised impacts from the WWTPs. This is consistent with Australian national GHG accounting data, which estimates the contribution of the "wastewater handling" sub-sector at 0.4% of the total national emissions (AGO, 2007). It seems obvious that policy or design considerations which are heavily focussed on GHG emissions, for political or other reasons, may be misdirected. However, without a life-cycle type approach this conclusion might not be intuitive. Indeed the problem of relating, comparing or objectifying different impacts is complex.

[23] Similar to Plants A and D at the two extremes of actual plants surveyed in this study in terms of size and effluent N & P concentrations – refer to Table 1.

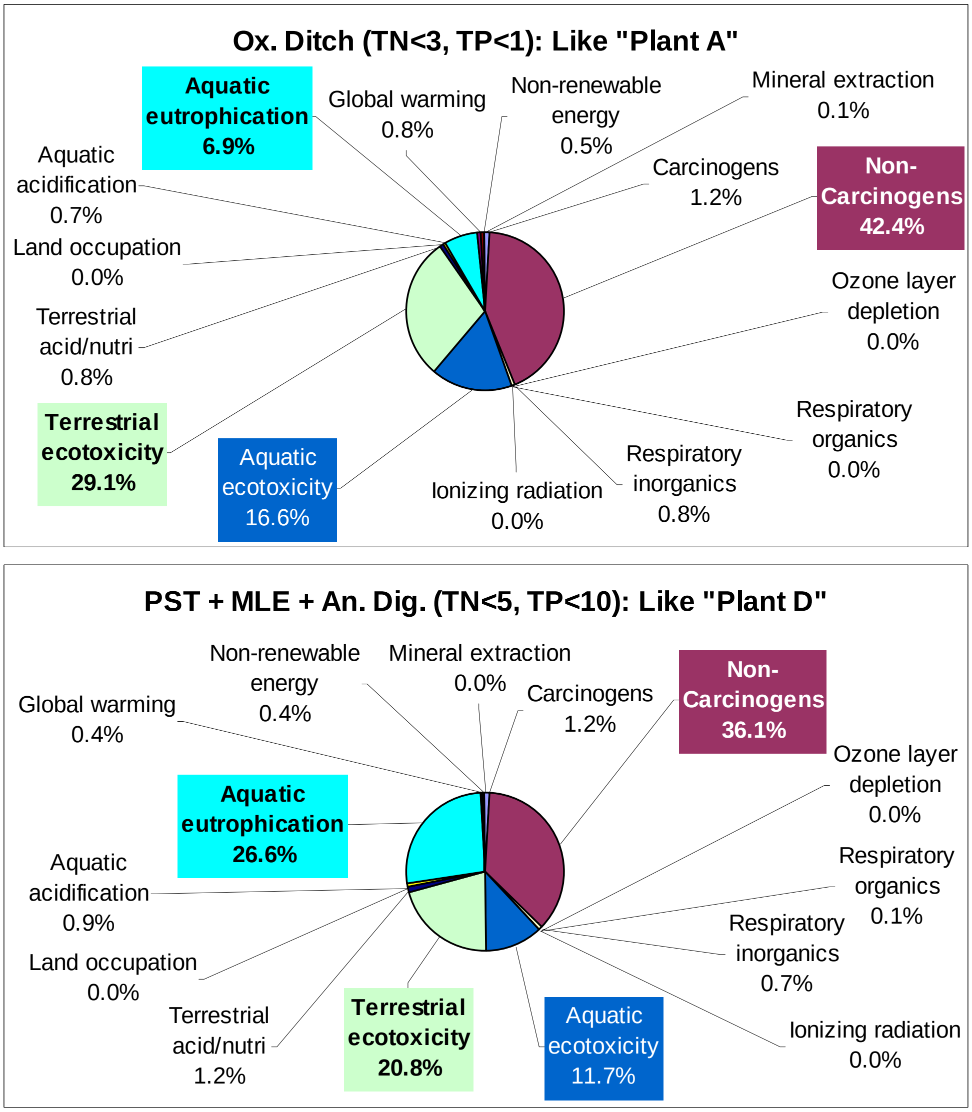

Figure 4 – Percentage contribution to sum of normalised impacts by LCIA category for two WWTP configurations based on data from the desktop study.

An LCA study of the type undertaken here, raises questions over the extent to which the dominant (normalised) impacts from a given process serving large populations, such as a WWTP, can (or should) be controlled by environmental regulation and appropriate design/ operation. Prevention of aquatic eutrophication is one of the prime reasons for building and operating a WWTP. This is clearly justified given its dominance in terms of normalised impacts (Figure 4). However, there are inevitable trade-offs in terms of increasing process intensity to achieve more advanced nutrient removal performance. These will typically include the following:

- Greater avoidance of artificial fertilizer use (and associated higher heavy metal/ non-carcinogen pollution) through greater nutrient recycling from biosolids containing more phosphorus when applied in agriculture *versus* impacts associated with transporting biosolids over greater distances from cities to farms *and* maximising the nutrient (N &/or P) content of the biosolids. Bio-P removal can be used to increase the P content of the biosolids, but this may pose process limitations for N removal that might require chemical supplementation with associated impacts (production, transport, GHG emissions etc.) or process design and operational constraints (see below). The cost and impacts of chemical supplementation for N removal will need to be compared against that of chemical supplementation for P removal, probably using metal salts containing iron or aluminium. Especially the latter has potentially significant impacts in the categories of aquatic or terrestrial ecotoxicity due to metal residuals reporting to either the receiving water or biosolids to land. This requires careful analysis in LCIA studies due to variability in metal bioavailability with soil types and conditions (e.g. application rate, pH etc.). Ideally a separate N and/or P-rich product (e.g. magnesium ammonium phosphate) should be produced at the WWTP as a means of recycling nutrients to minimise both artificial fertilizer use and impacts from transport of biosolids containing relatively low nutrient concentrations. The concentrated liquors from anaerobic digesters are attractive in that regard.

- Additional imported power inputs (GHG emissions) for aeration, mixing and recycling *versus* achieving greater biological nutrient (N & P) removal (BNR) in activated sludge plants (e.g. adopting extended aeration-type processes vs. primary sedimentation plus anaerobic digestion in the design of such plants). Given the apparently *relatively* major impact from eutrophication compared with the minor contribution of Global Warming (GHG emission) to total impacts from WWTPs, the move towards BNR designs in recent decades appears to be justified. If this is coupled with artificial fertilizer avoidance through agricultural use of P-rich biosolids (see above), total impacts can be further reduced.

The sum of the normalised impacts may be considered to an index of the Total Environmental Burden (TEB) posed by the WWTP over its life cycle, relative to that arising from all other anthropogenic activities of the population served. On that basis, in percentage terms, Figure 5 shows that the WWTP considered in the desktop study account for roughly 2 to 5% of the TEB. Alternative WWTP designs, with radically different nutrient removal capabilities or performance were compared in the desktop study. Some of these are shown in Figure 5 ranging from basic treatment on the right (high effluent total N) to virtually the limit of current technology (effluent TN<3 mg/L) on the left. The actual plants surveyed here for operating data (Plants A-D) all fall to the left, since all have advanced N removal at least.

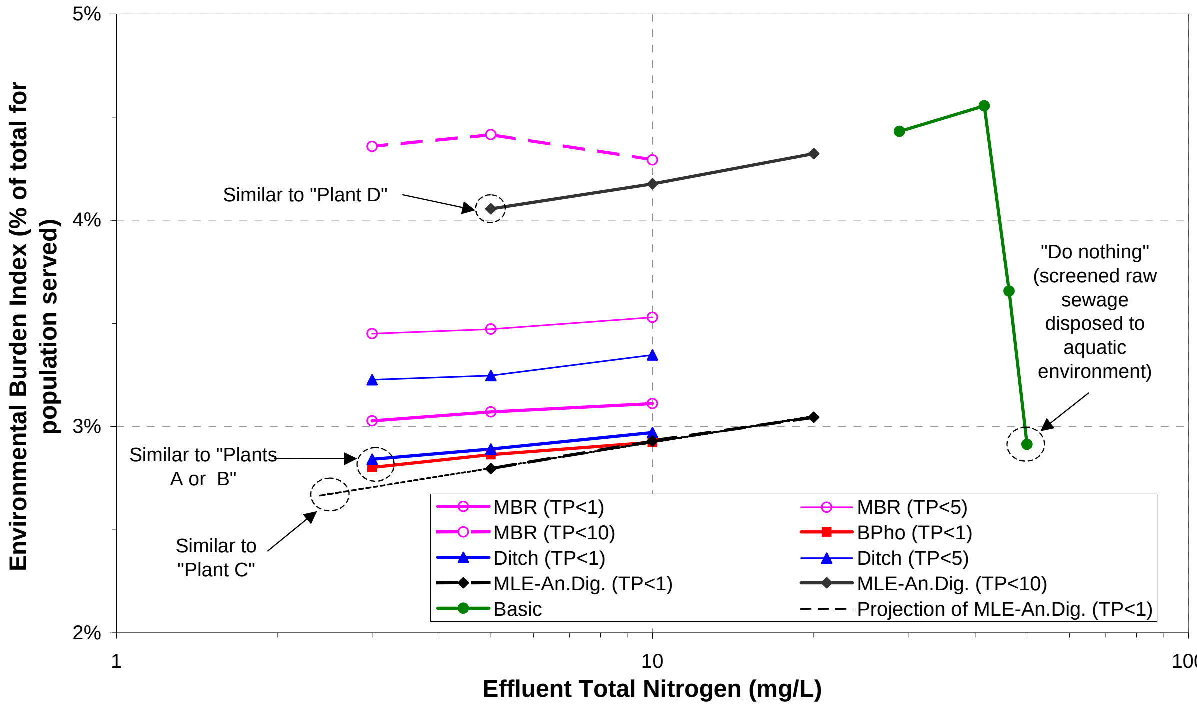

Figure 5 – Sum of normalised impacts (*EQUAL* weighting of all categories) from LCIA study (expressed in percentage terms as Total Environment Burden index) for WWTPs of different configurations in an Australian context.
MBR = Membrane Bioreactor; BPho = modified Bardenpho; Ditch = Oxidation Ditch; MLE = Modified Ludzack-Ettinger; An.Dig. = Anaerobic Digestion; Basic includes plants with Preliminary, Primary treatment or Aerobic (nitrification) only treatment. Effluent TP given in parentheses. Notes refer to actual WWTPs surveyed (see Table 1).

The question of weighting

In Figure 5 all LCIA categories were equally weighted. From the results in Figure 5 it appears that there is only a marginal reduction in TEB index when moving towards more advanced forms of wastewater treatment that achieve greater degrees of nutrient removal. In fact, from this assessment (equal weighting in categories), it may be argued that the most basic forms of treatment (or even the "Do Nothing" case where screened raw sewage is discharged to the aquatic environment) achieve approximately about the same overall TEB as the most advanced BNR plant. This result is surprising and likely to be highly contentious since it poses socio-economic and political questions in relation to environmental regulations. To what extent would communities be prepared to trade-off local deterioration of receiving water quality in the interests of minimising global impacts?

In order to explore this argument, a different weighting may be applied to aquatic eutrophication, relative to the other LCIA categories. For example, Figure 6 shows the same set of LCIA results for the WWTP configurations depicted in Figure 5, except that the weighting has been shifted to *five times higher* for eutrophication, relative to the other impact categories. In other words, the TEB is now calculated as the sum of the *unequally* weighted impacts from the WWTP (normalised to Australian national data for the population served) in each of the fifteen categories. Eutrophication now accounts for 27% and 64% of the attributed total impacts for WWTPs configurations similar to Plants A and D respectively (compare with Figure 4). For the

"Do Nothing" case (see above), eutrophication accounts for 95% of the total impacts when using this unequal weighting.

The profile in Figure 6 shows decreasing apparent TEB index in response to lower effluent Total N (or Total P) for selected WWTP configurations. This profile might be considered to be more consistent with typical current Environmental Protection Agency regulatory policy in Australia and many countries where advanced nutrient removal is required. It suggests that LCA is a powerful tool to assist decision makers (including managers, politicians and regulators) with policy formulation. It provides a useful tool for evaluating the extent to which environmental impacts from processes in the water industry (or any other industry) are effectively traded off by implementing regulatory policy. Ultimately, decisions on the policy formulation will be driven by a combination of political (including emotional) and socio-economic drivers. Damage (or end-point) models, as part LCA studies, aim to explore such scenarios and would be a logical extension of the work carried out this study.

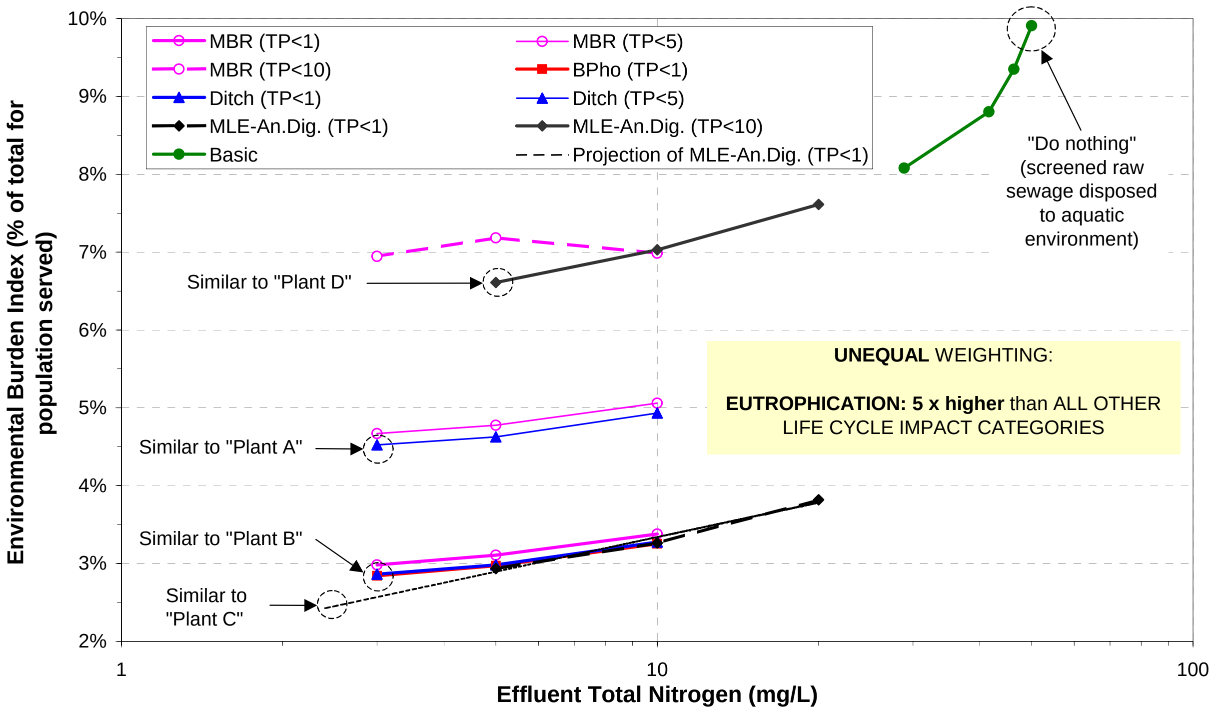

Figure 6 - Sum of normalised impacts (*UNEQUAL* weighting) from LCIA study – refer to text box and legend of Figure 5 for details.

CONCLUSIONS

The following conclusions may be drawn from this study:

1. Emission of Greenhouse gas (global warming) is a strongly emerging political issue in Australia. Current (or past) guideline methodologies published by federal government bodies for wastewater treatment plants (WWTPs) have been found to be inadequate. The best alternative identified in this study is one based from first principles on inventory data from WWTPs. However, provided sufficient data is available from benchmarked real operations, "rules of thumb" (e.g. tonnes of CO_2-e per ML treated) may be adequate.

2. Even using the best available data from literature sources, large uncertainties (in the order of approximately 10% to more than 100%) exist in the estimation of GHG emissions from WWTPs. These uncertainties are particularly due so-called “fugitive” emissions of nitrous oxide and methane from either the raw sewage entering the plant, secondary off-gases, or treatment and disposal of biosolids. Further research to better quantify these emissions is required.

3. Assuming relatively low “fugitive” direct emissions of nitrous oxide and methane, greenhouse gases from a WWTP are typically dominated by emissions due to imported electrical power, followed by those associated with biosolids treatment and disposal. Again, published data on emissions of nitrous oxide and methane from biosolids disposal operations such as landfilling, agriculture application (or other processes such as composting) show a wide range, which adds to uncertainties when reporting GHG emissions. More attention to these aspects will be required in order to establish consistent norms or guidelines for national (or international) reporting systems in the lead up to increased carbon trading globally.

4. WWTPs with primary sedimentation plus anaerobic digestion offer the best option currently available for reducing GHG emissions, even if followed by biological nutrient removal (BNR) processes that require supplementary addition of chemicals for low N & P effluents. Compared to extended aeration-type BNR plants widely adopted in recent years, reductions in the order of 20 to 40% appear to be feasible with anaerobic digestion and power generation from biogas. The same conclusion has emerged from both desktop assessments and actual WWTP operations in SE Queensland (Australia).

5. Good overall agreement was achieved between GHG emissions estimates based on actual WWTP operating data and those carried out independently using a desktop modelling study, which formed the basis for a Life Cycle Assessment approach to comparing alternative WWTP process configurations. Although the methodologies for GHG calculations were similar, some differences in approach or actual vs. model data were identified. Nevertheless, the desktop study was validated to some extent by benchmarking against real operating data.

6. From the desktop LCA study, it was noted that impacts from a WWTP, when normalised against an Australia-wide inventory, are dominated by eutrophication and the effects of either heavy metals (or light metals such as aluminium) that either accumulate in the biosolids disposed to terrestrial environment or have dissolved residuals, which may be released to the aquatic environment. Greenhouse gas (global warming) impacts from the WWTP are relatively minor when normalised against Australian national data. This raised the question of whether the current focus on GHG emissions at WWTPs is misdirected.

7. The LCA study also showed that in terms of aggregated, normalised impacts (i.e. so-called Total Environmental Burden) with equal weighting for all impact categories, more advanced WWTPs do not necessarily have a lower environmental burden. This raised the question of whether regulations directed at improving nutrient removal performance of WWTPs are justified. Nutrient recycling might help further to reduce total environmental burden but might still not produce a significant improvement compared to the “Do nothing” option of disposing screened raw sewage to receiving waters.

8. It was concluded that generalised life cycle impact assessments can only inform but not direct decision-making in respect of environmental regulation. Policy decisions in this regard are made in a complex milieu, which includes many socio-economic and political factors. These reflect societal values and may be strongly subjective. For example, to align the profile of Total Environmental Burden index from life cycle impact assessment data with current environmental policy driving nutrient removal WWTP design (lower effluent N &/or P), eutrophication must be weighted five times higher than all other impact categories. This illustrates the usefulness of LCA as a tool to inform policy or decision making in an effort to embrace sustainability on the widest possible front.

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