Modeling the Carbon Footprint of Various Biosolids Treatment Options

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ABSTRACT

There are a wide variety of processes which are capable of treating biosolids to comply with the pathogen reduction requirements defined by US EPA 503 regulations. Choice of technology is complex and dependent on a number of project specific and economic influences. However, within Europe, long term sustainability and carbon impact are becoming more influential and this is being backed up by economic incentives in the form of taxes and levies. This paper presents results of a model developed to calculate the carbon footprints of a number biosolids treatment processes. Results showed that Class A systems were more carbon intensive than Class B equivalents due to additional power required to meet pasteurization temperatures. Generally, processes with anaerobic digestion had far smaller carbon impacts due to lower quantities of biosolids for downstream processing and transport and the production of renewable energy. The benefits of fertilizer displacement by composting were also significant.

KEYWORDS: Biosolids treatment, carbon footprint, anaerobic digestion.

INTRODUCTION

It has eventually and sometimes reluctantly, become understood that climate change is directly influenced by the anthropogenic release of what are known as "Green House Gases – GHGs". Their release into the earth's atmosphere has been correlated with an increase in global temperature which in turn, has been linked to weather change with potentially catastrophic impacts (IPCC, 2001). With such significant global impacts expected, it is therefore no surprise that a great deal of interest is being shown by industry and the media alike on highlighting the issues and also proposing methods of reducing the generation of GHGs.

GHGs have been classified by the Intergovernmental Panel on Climate Change (IPCC), and include: carbon dioxide; methane; nitrous oxide; tetraflouromethane; sulfur hexafluoride; HCFs; CFCs, and others. Determination of carbon footprint involves measuring the amount of these compounds emitted during a process and converting the data to a carbon dioxide equivalent. The carbon dioxide equivalent, (also expressed as CO_2eq or CO_2e), is an internationally accepted factor that expresses the global warming potential a GHG has in terms of the quantity of carbon dioxide that would have an identical impact. For example, the CO_2e for methane is 21 (IPCC, 2001) implying that 1 unit of methane has the same global warming potential as 21 units of carbon dioxide. There is an implicit link between power consumption and carbon footprint, consequentially the power sector is responsible for 37% of all anthropogenic CO_2 (Ashton, 2005). In the UK, the Water Industry accounts for 3% of all power consumed (Water UK, 2007). Most of this power is used to run processes and pump water. The impact of power on carbon footprint of United Utilities, a UK Water Company, is shown in Figure 1.The power demand of the Water Industry is partially offset by the generation of a methane-enriched biogas – produced during the anaerobic digestion of sewage biosolids – to electricity and heat via combined heat and power plants.

Biosolids Treatment

Power is required to treat biosolids to an acceptable level before use. As mentioned in the previous section, anaerobic digestion can be used as a process step and this may reduce overall power requirements. By law, biosolids must be treated to reduce the risk from pathogens and vectors prior to their application to or placement on the land for beneficial use or disposal (US EPA, 1999). Treatment types fall into two categories: Processes to Further Reduce Pathogens (PFRP), known as Class A; and, Processes to Significantly Reduce Pathogens, labeled Class B. The attainment of Class A quality is the more stringent of the standards and requires:

- a density of fecal coliforms in the sludge of less than 1000 most probable number (MPN) per gram of total dry solids, **or**
- a density of Salmonella sp. bacteria lower than 3 MPN/4 g of total dry solids, **and**
- Time/temperature relationship according to various equations

These standards apply to the point at which the biosolids are used. Class A can be attained via numerous treatment systems such as: composting, heat drying, heat treatment, thermophillic anaerobic digestion, β and γ radiation, and pasteurization (70°C for 30 minutes). Time temperature relationships required for the pasteurization of various micro-organisms have been well documented in the work of Feacham *et al*. (1983).

The requirements for Class B, or Processes to Significantly Reduce Pathogens (PSRP), are lower and therefore Class B treated biosolids have more restrictions on application. Here, biosolids quality must have a geometric mean fecal coliform density of < 2,000,000 colony forming units (cfu) or most probable number (MPN) per gram of solids dry weight. Class B systems include: aerobic digestion; mesophilic anaerobic digestion (MAD); composting; lime stabilization and air drying.

United Utilities

United Utilities owns and operates the water and wastewater networks and also operates electricity distribution, in the North West of England, UK. The company manages over 700 water and wastewater treatment works, 32,000 electricity sub-stations and around 120,000 kilometers of water pipes, sewers and cables. It is a major service provider in the competitive industrial water and wastewater market and is responsible for the relationship with 3.1 million household and business accounts serving 7 million customers. Based on recent records, United Utilities produces approximately 213,000 tones biosolids per year. The vast majority (over 70%) of these biosolids are digested and primarily recycled to land. The remaining biosolids are raw and treated with lime prior to land application. A small fraction (12%) of the biosolids are incinerated in a purpose-built plant in Widnes.

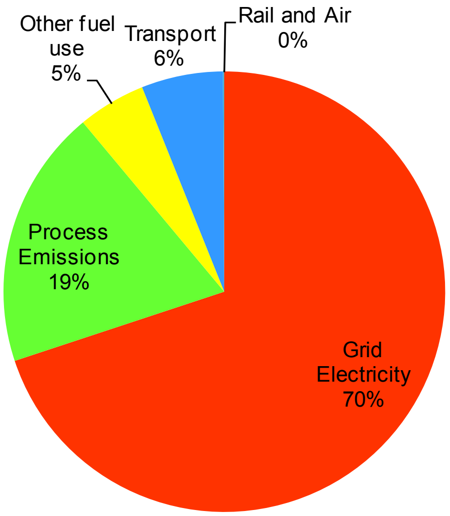

Figure 1. Composition of United Utilities' carbon footprint

The latest estimate of United Utilities' carbon footprint puts it at 401,500 tones CO_2e/year as owned by the company. It is further estimated that its actions influence a further 1.66 million tones CO_2e/year. The influenced carbon dioxide data refer to those generated by suppliers, third parties working with United Utilities etc.

Figure 1 shows the composition of United Utilities' carbon footprint. It is clearly evident that process technologies play a vital role and contribute nearly 90% of the total (70% from power and 19% from emissions). Therefore, any strategy to reduce carbon footprint will have to involve process optimization and measures to further reduce power consumption, for example additional implementation of anaerobic digestion. However, increasingly strict legislative targets being imposed on the Water Industry in the UK are causing the Water Industry's power demand to increase. According to Marsh (2002), tightening a BOD concentration limit from 25 to 5 mg/l increases energy demand by three times, and increases the emission of GHGs by a factor of five. The author suggests that a trade-off exists between effluent quality and energy requirement.

There is growing interest regarding the impact of carbon footprint in the UK Water Industry in spite of the tightening consent requirements. The Office of Water Services (OFWAT), the Water Industry Regulator, has now made it statutory for Water Companies to disclose operating and embodied carbon (that generated during the manufacturing and maintenance of the plant) footprints of new proposed solutions.

Calculating carbon footprint in the UK Water Industry

Generally, the vast majority of the work involved in determining carbon footprints entails calculating the inputs required for carbon modeling. The inputs (e.g. power; transport; chemical consumption; quantity and type of building material used) are then multiplied by emission or conversion factors. The most well-known attempts to determine a Water Industry carbon footprint in the UK were made by UKWIR, which has developed a model entitled "Workbook for Quantifying Greenhouse Gas Emissions" (2005). The model includes Water Industry-specific data involving emissions of GHGs from wastewater treatment and application of sewage biosolids to land. The model has been the foundation for further work on carbon calculating (Entec UK, 2006). Since then, UKWIR is in the process of updating the workbook with revisions and additions, such as inclusion of embodied carbon calculating. This work is currently ongoing.

United Utilities' first work on carbon calculating was to assist in the development of a sustainable biosolids strategy. The original model was developed by United Utilities and MWH and was used to determine the long term sustainability of a biosolids strategy for 52,000 tones dry solids of biosolids. The existing solution of liming biosolids prior to land application was not considered sustainable. As reliance on land was considered high risk due to legislation and many other external factors, a biosolids incinerator was proposed. Since the proposition several years ago, the drivers for the project had changed and a new strategy was eventually developed which involved the installation of advanced digestion at a digestion plant which feeds United Utilities' incinerator. The solution allowed flexibility regarding the ability to either apply advanced treated biosolids to land, or send the biosolids down a pipeline for incineration. Use of the model to calculate carbon footprints played a pivotal role in the development of the new strategy. The results were presented to the Environment Agency and OFWAT who gave their support. The result has been a fundamental change to a $250 million biosolids program and a reduction in United Utilities' carbon footprint of 32,500 tones CO_2e/yr. This is a reduction of 8% based on today's figures.

Since that work, the model has been updated and comprehensively expanded to be more generic, include a greater number of process technologies and incorporate more internal calculations thus reducing the quantity of assumptions required. This paper presents results from model simulations for processes which are capable of achieving either Class A or Class B quality of biosolids as defined by US EPA 503 Regulations.

METHODOLOGY

There are two fundamental differences between the current model presented here and previous attempts described above at calculating biosolids carbon footprints. The first difference is that it adds a layer of calculation which performs all the necessary mass and energy balances – previous models required the outputs of a mass and energy balance as an input, i.e. input kilowatt hours consumed. This allows inputs to be simpler, for example entering the throughput, BOD and ammonia concentrations and expected dry solids output from a dewatering device, the model will automatically calculate the power required for dewatering and liquor treatment and the chemicals, (both polymer and alkalinity for total nitrogen removal) and then convert them into a carbon footprint. The second difference is that the model has been designed to easily benchmark whole solutions, for instance all biosolids processing at the sewage works, transport of biosolids from the works and application of the biosolids to land, rather than individual process unit items. It calculates different biosolids solutions based on the same basic inputs which allows a quick comparison on which outlet to pursue for a particular project. Other more detailed differences are described elsewhere (Barber, 2008).

Central to the model is the impact of anaerobic digestion on down-stream processing. For mesophilic anaerobic digestion (MAD), a volatile solids destruction figure is input into the model which then calculates the biogas generated; number of CHP engines required and surplus waste heat recovery potential (both hot water and steam) from the CHP plant after space heating and convective heat losses from the digestion plant have been accounted for. Based on external conditions and temperature, it calculates additional heat requirements – if required – for TAD, liquid pasteurization and ATAD. The additional heat required for ATAD is calculated after exothermic heat release has been determined based on the strength of the biosolids and hydraulic retention time. For thermal hydrolysis, the model determines the steam required for the pre-treatment plant and compares it to the steam available from the CHP plant. It then redirects biogas from the CHP plant towards a boiler to make up the deficit in energy requirement. Finally, for MAD combined with drying, the model can be run to redirect biogas into the dryer or to use biogas for energy generation and use natural gas for drying.

Much in the same way that biogas can be used to displace fossil-fuel derived energy, a fertilizer benefit was calculated based on the fertilizer value of the biosolids and the energy required to manufacture an equivalent quantity of fertilizer (Kroiss and Zessner, 2007). In this study, it is assumed that only half of the displacement benefit is counted. The model is also set up to account for differences in crop yields between fertilizer and biosolids to include for sequestering effects. However, in this paper, it is assumed that crop yields are similar for both.

The model calculates an operating carbon footprint by subtracting the carbon footprint benefits (of renewable energy generation from biogas; energy generation from biosolids incineration; and displacement of fertilizer) from the factors which contribute to carbon (such as power consumption; chemical usage and biosolids transport). For example, for Class A liming of 50,000 tones biosolids/yr the contributing carbon footprint [comprised of dewatering and liquor treatment (power and chemicals); power to run liming plant; lime; biosolids transport and land emission] is approximately 49,500 tones CO_2e/yr. The benefits of displacing an equivalent amount of fertilizer are a saving of 9,800 tones CO_2e/yr from not burning fossil fuels. Therefore, the operating carbon footprint is considered to be 49,500 – 9,800 = 39,700 tones CO_2e/yr.

The results presented in the remainder of this manuscript are based on that method of calculation, and other features of the model are described later in relevant sections.

Operating carbon footprints were determined for a number of Class A and Class B processes defined in Figure 2 and Table 1.

All carbon footprints results are based on a plant processing 50,000 tones dry biosolids per year. The carbon footprints presented in this paper are all based on land application of biosolids apart from incineration. Other general assumptions are presented in Table 2.

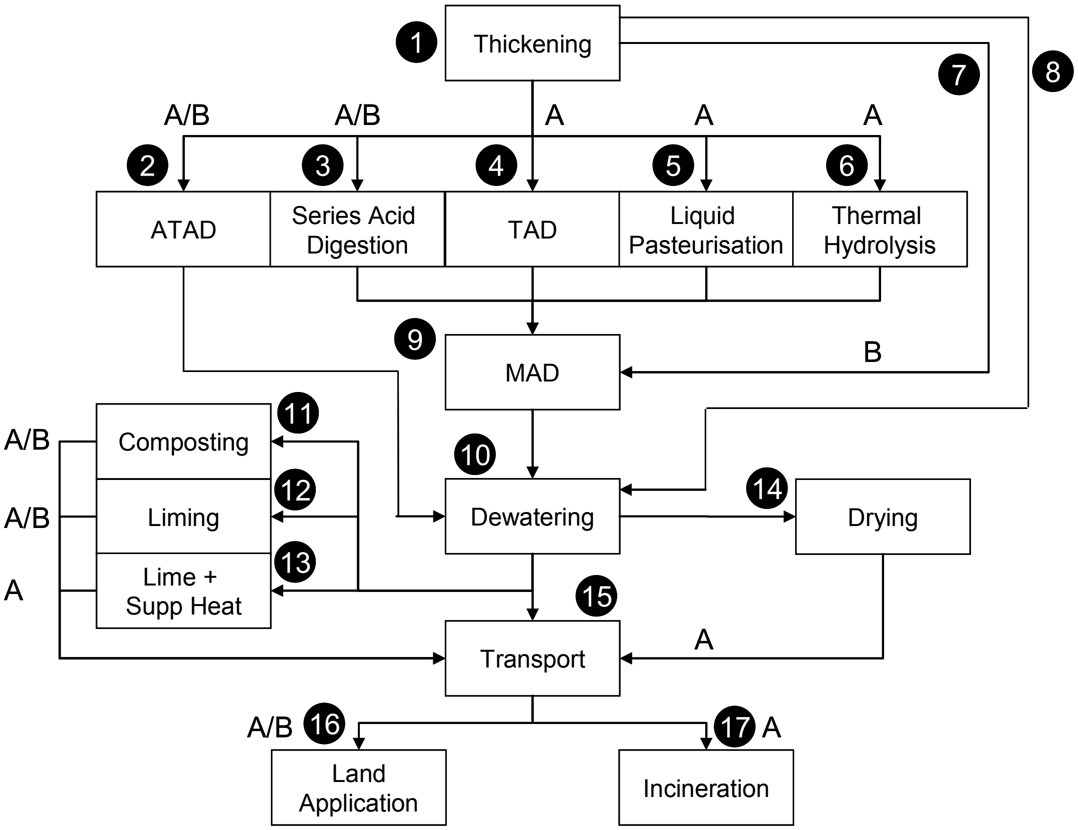

Figure 2. Processes evaluated in carbon footprint model. Numbers refer to process sequence required. For example, liquid pasteurization followed by MAD and land application comprises steps: 1 (thickening), 5 (liquid pasteurization), 9 (MAD), 10 (dewatering), 15 (transport) and 16 (land application). (For thickening and dewatering, liquor treatment is included in the calculations but excluded from diagram for simplicity). Letters refer to processes evaluated based on achieving Class A, B, or both as defined in US EPA 503 regulations for treatment of biosolids.

Table 1. Processes' evaluated for carbon footprint

Process	**Process Steps** [a]					
CLASS A SYSTEMS						
Advanced digestion + incineration	1	6	9	10	15	17
Autothermal aerobic digestion (ATAD)	1	2	10	15	16	
Composting	1	8	11	15	16	
Lime + supplemental heat	1	8	13	15	16	
Liming	1	8	12	15	16	
Liquid Pasteurization + MAD	1	5	9	10	15	16
MAD + drying	1	7	10	14	15	16
MAD + incineration	1	7	10	15	17	
Raw biosolids drying	1	8	14	15	16	

Process	**Process Steps [a]**					
Raw biosolids incineration	1	8	15	17		
Series acid phase digestion + MAD	1	3	9	10	15	16
Temperature phased digestion TAD + MAD	1	4	9	10	15	16
Thermal hydrolysis + MAD	1	6	9	10	15	16
CLASS B SYSTEMS						
Autothermal aerobic digestion (ATAD)	1	2	10	15	16	
Composting	1	8	11	15	16	
Liming	1	8	12	15	16	
Mesophilic Anaerobic Digestion (MAD)	1	9	10	15	16	
Series acid phase digestion	1	3	9	10	15	16

[a] For use with Figure 2

Table 2. General Assumptions for all processes

Parameter	**Data**	**Units**	**Notes**
Biosolids Processed	50,000	[TDSA]	
Dry Solids in thickener	3%	[DS]	
Raw biosolids volatile matter	75%	[VS]	
Dry Solids entering digestion	6%	[DS]	For series acid phase; entering TAD; ATAD; Pre-MAD pasteurization
	10%	[DS]	For thermal hydrolysis + MAD
	4%	[DS]	(Calculated) entering MAD after TAD
Dry Solids from dewatering	25%	[DS]	For raw and digested biosolids
	33%	[DS]	For thermally hydrolyzed digested biosolids
Polymer consumption	8	[kg/TDS]	
BOD Concentration	3000	[mg/l]	For raw biosolids
	2250	[mg/l]	For digested biosolids
Ammonia Concentration	100	[mg/l]	For raw biosolids
	980	[mg/l]	For digested biosolids
	2500	[mg/l]	For thermally hydrolyzed biosolids (Fjaergard *et al.*, 2006)
Transport	25	[miles]	For biosolids cake applied to land; green waste for composting
	30	[miles]	For biosolids cake to incineration plant
	40	[miles]	For dried biosolids applied to land

Other, more process specific assumptions and model inputs are given in Tables 3 to 7 below:

Table 3. Assumptions for liming processes

Parameter	Data	Units	Notes
Lime addition	60	[%]	On dry basis for Class A liming
	20	[%]	On dry basis for Class B liming, and for Class A lime + supplemental heat system[a]
Power for liming plant	30	[kWhr/TDSA]	Calculated from data in Harrison, 1993
Dewatering improvement	3	[% points]	From site data (Barber, 2002)

[a] Additional heat requirement calculated after heat generation from lime addition accounted for

Table 4. Assumptions for ATAD

Parameter	Data	Units	Notes
Hydraulic Retention Time	16	[d]	For Class A
	40	[d]	For Class B
Heat Release	29.075	[kWhr /m^3biosolids]	For Class A or B; calculated from Metcalf and Eddy 4th Edn (2004)
Mixing energy	0.026	[kWhr /m^3]	From Metcalf and Eddy 4th Edn (2004)

Table 5. Assumed volatile solids destruction for digestion processes

Process	Data	Units	Notes
Class B ATAD	42	[%]	Calculated based on temperature and HRT
Standard MAD	45	[%]	Based on average UK performance
Class A ATAD	45	[%]	Calculated based on temperature and HRT
Class A series acid digestion + MAD	50	[%]	Werker *et al.*, 2007
TAD + MAD	55	[%]	Wilson *et al.*, 2004
Class B series acid digestion + MAD	58	[%]	Hoyland *et al.*, 2006
Thermal hydrolysis followed + MAD	60	[%]	Hoyland *et al.*, 2006

Table 6. Assumptions for Drying Processes

Process	Data	Units
Dry solids exiting dryer	92	[%]
Thermal Energy need – Gas	3500	[MJ/t water evaporation]
Electrical Requirement	60	[kWhr/t water evaporation]

Table 7. Assumptions for Biosolids Incineration

Process	Data	Units	Notes
Biosolids carbon content	55	[%]	Raw biosolids
	53	[%]	Digested biosolids
Biosolids nitrogen content	10	[%]	
Power generation efficiency	15	[%]	
Downtime	1	[months]	For planned maintenance. During shut-down it is assumed that biosolids are applied to land (with lime if raw)
Biosolids Lower Heating Value (LHV)[a]	17000	[MJ/kg DS]	Raw biosolids
	13000	[MJ/kg DS]	Digested biosolids
	12000	[MJ/kg DS]	Thermally hydrolyzed biosolids
Chemicals	0.0458	[t/t biosolids incinerated]	Total chemicals, calculated from WID compliant incineration design model
Start-up gas	0.697	[m^3/tDS incinerated]	Based on plant data

[a] MJ/kg DS = BTU/lb x 429.9

Note: this paper only shows results of operating carbon footprints and not those associated with the carbon release during manufacture and maintenance of the plant – known as embodied carbon. Detailed work on a number of solutions for embodied energy was performed in the study described earlier for determining the new biosolids strategy. It was found that over a period of 30 years operation, the construction and maintenance carbon footprints of the solutions (for large construction projects – e.g. a raw biosolids incineration plant) contributed typically less than 2% of the whole life carbon, implying that yearly operation of the plant was far more significant with respect to carbon footprint than its construction.

RESULTS

Class A Processes

Figure 3 shows the contributions and benefits to carbon footprint for the Class A options determined in this study.

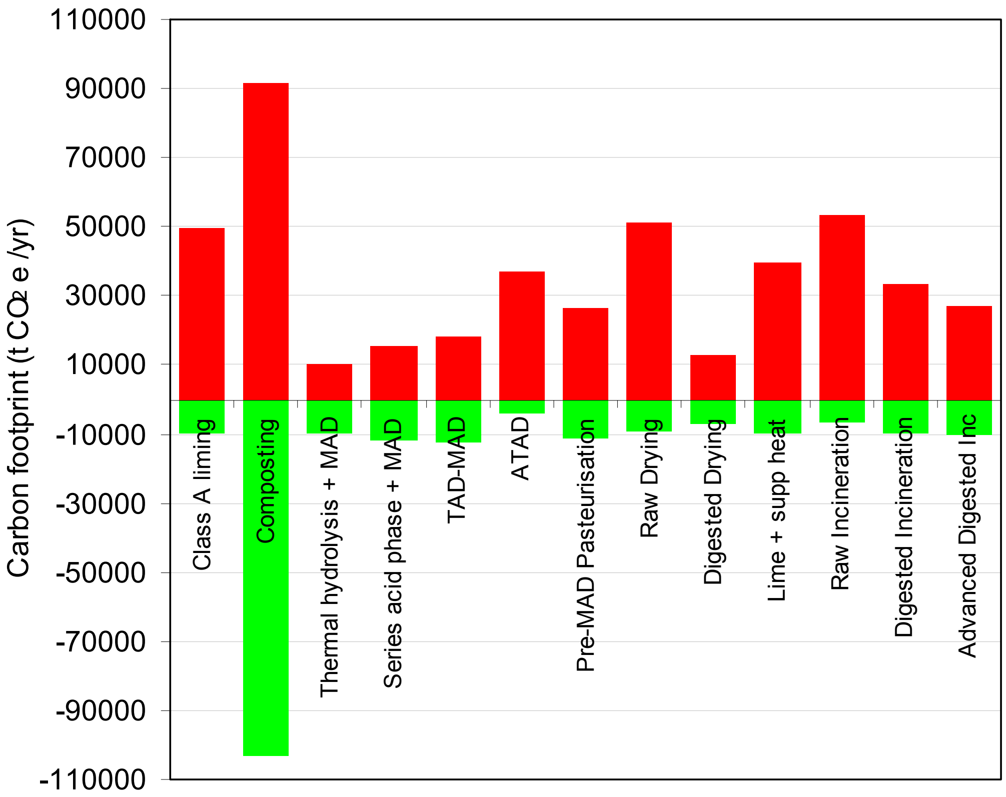

Figure 3. Carbon footprint contributions (🟥); and benefits (🟩) for Class A compliant processes. Note: operating carbon footprint calculated as sum of contributions minus benefits.

Figure 4 shows the overall carbon footprints (i.e. the sum of contributory factors minus benefits). Generally, processes which do not generate renewable energy in the form of biogas, such as raw biosolids options (red columns in Figure 4), and processes which consume large quantities of power (for example, drying, ATAD and incineration) are the most carbon intensive. The lowest carbon footprint coincides with composting, if; the benefits of fertilizer production displacement are counted. For the compositing process, the quantity of green waste (with C:N ratio 30:1 and dry solids 50%) was determined to enable ideal conditions for composting with biosolids (Outwater, 1994).

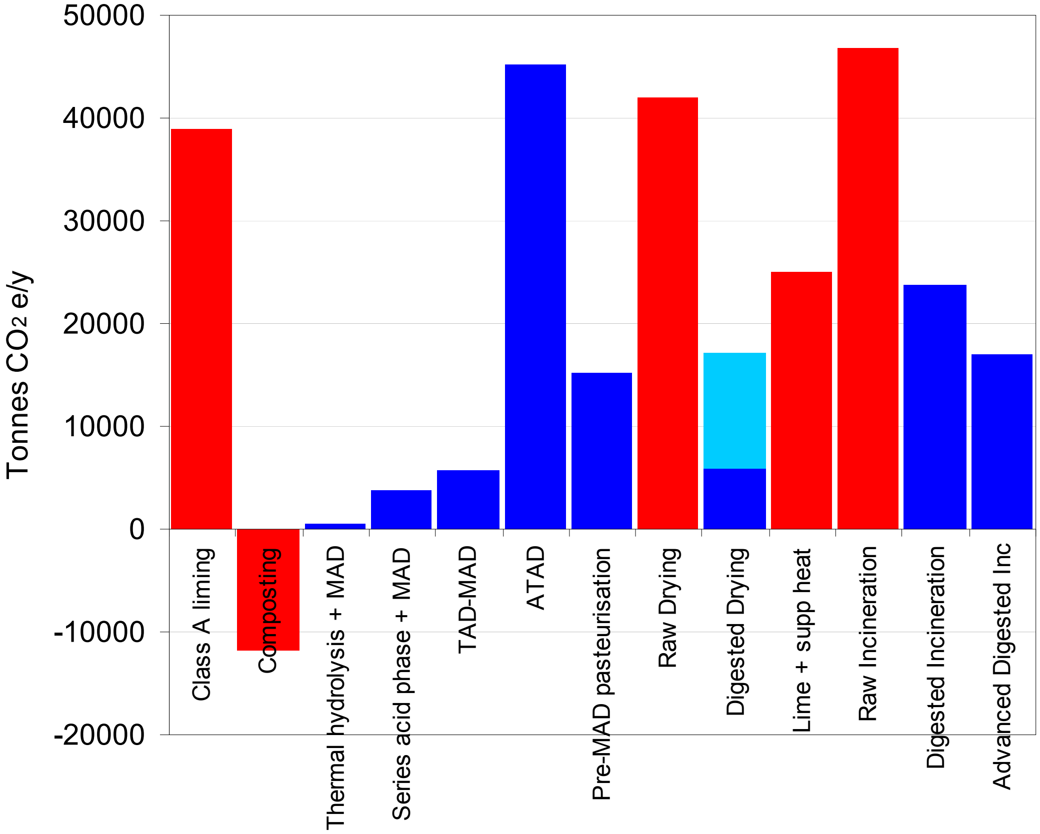

Figure 4. Operating carbon footprint for Class A compliant biosolids treatment processes. Key (🟥 processes without digestion;🟦 processes with digestion) Note: for digestion + drying the height of the darker blue bar is the carbon footprint when biogas is used for drying. The height of lighter blue bar is the carbon footprint when biogas is used for electricity generation and natural gas is used for drying

The benefits for composting biosolids are especially large due to fertilizer displacement from, not only the biosolids (as would be the case for limed biosolids to land) but also the green waste added to it. Otherwise, composting generates a large carbon footprint as shown in Figure 3. Figure 5b) shows that the vast majority of the carbon footprint produced from composting – nearly 50% – comes from transporting the large quantities of green waste required to meet optimal dry solids and carbon:nitrogen ratios required. Aeration contributes under 15% of the footprint in comparison.

After composting, the lowest carbon footprints were associated with land application of advanced digested biosolids with the following technologies – in order of lowest carbon footprint first – thermal hydrolysis (Panter, and Kleiven, 2005), series acid-phase digestion – known as Enhanced Enzymic Hydrolyis (EEH) in the UK (Le *et al.*, 2006) and thermophilic digestion (Wilson, *et al.*, 2004). [Note: in this study, the influence of potential pathogen re-growth following centrifugation (Monteleone *et al.*, 2004), has not been considered. Processes which exhibit re-growth will require further treatment which would increase carbon footprint].

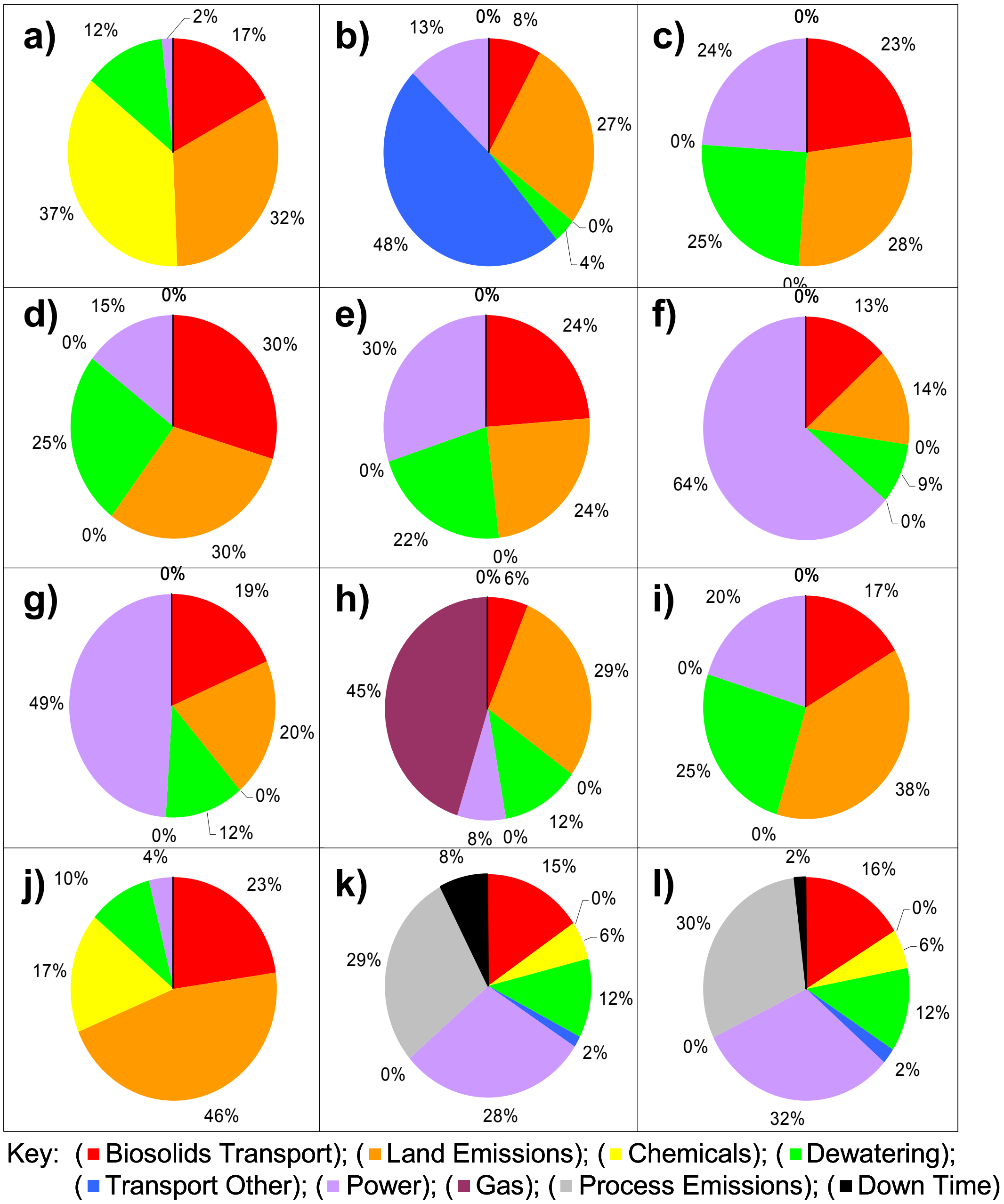

Figure 5. Composition of carbon footprint for Class A compliant biosolids processes. a) Liming; b) Composting; c) thermal hydrolysis + MAD; d) biological hydrolysis (acid-phase) + MAD; e) TAD-MAD; f) ATAD; g) Pre-digestion liquid pasteurization + MAD; h) Raw biosolids drying; i) Digested biosolids drying; j) Lime + supplemental heat of raw biosolids; k) Raw biosolids incineration; l) Digested biosolids incineration.

Generally, advanced digestion processes generated low carbon footprints due to a combination of: renewable energy generation; lower biosolids transport (than raw process alternatives) and lower emissions of methane and nitrous oxide when applied to land (UKWIR, 2005). In terms of renewable energy generation, there was little difference between the technologies. Thermal hydrolysis has been shown to generate more biogas than other similar processes (Hoyland *et al.*, 2006) however; it has a parasitic demand of biogas to generate process steam. This study found that the parasitic demand cancelled out the increased generation, making it no better or worse than the other pre-treatment technologies with respect to energy generation. However, with respect to carbon, thermal hydrolysis still had a major advantage (as shown in Figure 4), and this was due to better dewaterability (Evans, 2006) which reduces carbon emissions from transport, and lower land application emissions as more biosolids had been converted to biogas, albeit not necessarily converted to renewable energy. The advantage of thermal hydrolysis also increases with processes of increasing complexity downstream of digestion, such as drying or incineration (discussed later), and also with increasing transport.

Pre-MAD liquid pasteurization to 70°C had a far higher footprint, similar to digestion with drying, or advanced digestion with incineration. This was due to a large energy demand which contributed nearly half of its footprint (Fig 5g), even though it was assumed that the biosolids were preheated with a heat exchanger to 45°C. In fact, if it weren't for the generation of biogas, the carbon footprint of this option would more than double from 15,000 t CO_2e to < 35,000 t CO_2e.

Of the digestion options, ATAD generated the highest footprint, which even rivaled that generated from raw biosolids incineration. With respect to carbon impact, ATAD loses the benefits of all the other digested options, of fossil-fuel displacement due to renewable energy generation. Furthermore, as it is a digestion process and destroys a quantity of biosolids, less biosolids are then applied to land (when compared to raw biosolids processes) resulting in a concomitantly lower fertilizer displacement benefit. Coupled to the lack of benefits, ATAD has a heavy demand for power for both aeration and mixing, even though heating requirements are negligible or non-existent due to the exothermic reactions involved. Power contributes two thirds of the carbon footprint for ATAD as shown in Figure 5f).

For options involving liming to pasteurize biosolids to Class A status, the lime with supplemental heat system, known as the RDP process (Harrison, 1993) was found to be more carbon-friendly than systems which rely on overdosing lime to achieve required pasteurization temperatures. The supplemental heat system generated a carbon footprint approximately a third lower than that generated for the lime-only equivalent as demonstrated in Figure 4. This was due to a combination of three factors. Firstly, the carbon footprint of the additional lime added to meet pasteurization temperature was greater than the carbon footprint generated from providing electricity to heat the biosolids by the same temperature increase. Lime has an inherently high carbon footprint due to the large quantity of energy required to separate it from calcium carbonate and the subsequent release of carbon dioxide during the separation process. The impact of lime is clearly shown in Figures 5a) lime-only and 5j) supplemental heat processes. For the lime-only process, lime accounts for almost 40% of the carbon footprint, but this falls to under 20% for the supplemental heat process. For the supplemental heat process, the carbon footprint of the lime can be further reduced by the use of other admixtures (Barber, 2000) which may be waste streams and therefore have negligible embodied carbon. Secondly, the larger quantity of lime required by the lime-only process adds more bulk to the biosolids which in turn increases GHG emissions from transport. Finally, plant data (Barber, 2002) has shown that a combination of lime and electrical heat can cause an additional amount of moisture to evaporate from the biosolids and this would further reduce emissions from transport.

Drying of raw biosolids generates over twice the carbon footprint when compared to drying biosolids in combination with anaerobic digestion (Figure 4). Firstly, this is due to having to dry more biosolids as none have been destroyed. This increases water evaporation and subsequent power requirement of the dryer. Secondly, there is no renewable energy generation from biogas. Power accounts for over half of the carbon footprint of raw biosolids drying as shown in Figure 5h). Carbon footprint of drying with digestion falls further if biogas is used for drying as shown in Figure 4. Figure 6 shows the impact of volatile solids reduction and use/non-use of biogas for drying. At 45% volatile solids reduction – typical of UK performance – 4.0 MW of electrical power is generated if the biogas is consumed in the CHP plant. This drops to 2.1 MW if the required quantity of biogas is diverted to meet the dryer gas need. This causes carbon footprint to fall. Sensitivity analysis of the data shows that the quantity of benefit derived from using the biogas reduces as VSR increases. This is due to a smaller energy demand but also to the difference in emission factors between fossil-fuel displacement by CHP and natural gas. When natural gas is burnt it emits 0.206 kg CO_2e/kWhr (Defra, 2007) during drying, but when converted to power via CHP it displaces an equivalent quantity of fossil-fuel generated power which emits 0.523 kg CO_2e/kWhr (Defra, 2007). By default, as VSR increases the carbon benefits from CHP also increase, so diverting biogas to the dryer will reduce the benefits of the whole solution.

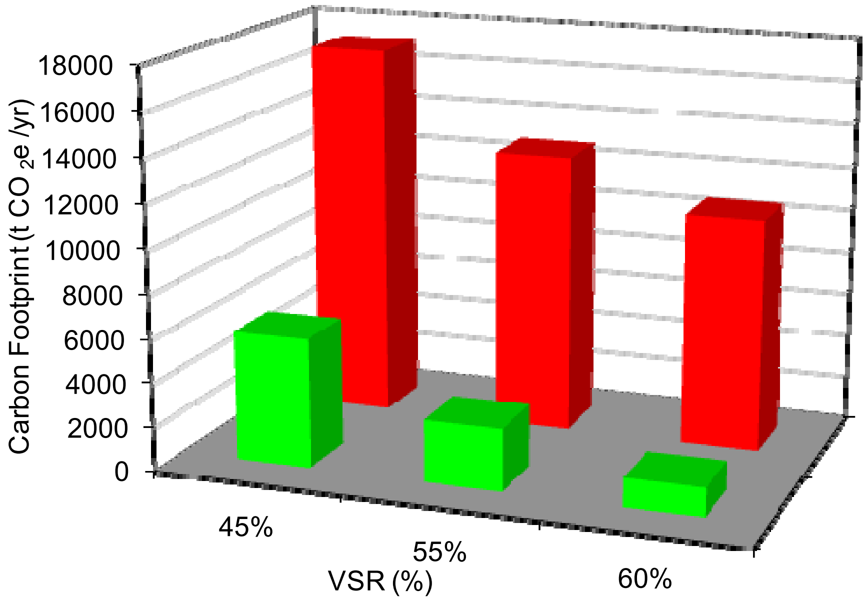

Figure 6. Influence of use (🟩); and non-use (🟥) of biogas and volatile solids reduction (VSR) on carbon footprint for MAD plus drying.

As one would expect, being a more complex process, incineration of biosolids involves a larger number of components to make up its operating carbon footprint as shown in Figure 5k) for raw, and 5l) for digested incineration. A similar pie chart was plotted for advanced digested incineration (based on thermal hydrolysis) but is not shown as it was not significantly different to Figure 5l). The carbon impact of incineration when preceded by anaerobic digestion is half that of when there is no digestion, even though more energy is generated in the incinerator plant for the raw option due to the (raw) biosolids having higher heating value. A holistic energy balance including both incineration and upstream digestion plants shows that more energy is recovered from the biosolids if digested first, even though generation of energy at the incineration plant is lower for digested biosolids. Additionally, the energy demand (hence carbon footprint) of the incinerator is far lower with digestion as the majority of operating requirements, such as combustion air, flue-gas scrubbing chemicals, natural gas, etc are all throughput influenced. If fewer biosolids are burnt – due to digestion – then the incinerator and its demand for consumables are inherently smaller. Such are the benefits of digestion prior to incineration of: better overall energy recovery, lower overall energy demand, lower emissions of GHGs (both in the incinerator and during down-time when biosolids are applied elsewhere), and lower operating requirements (all resulting in a lower carbon footprint), that there is no technical reason why a new incineration plant would be built without digestion upstream of it. When thermal hydrolysis precedes the digestion plant, the benefits increase further and the carbon footprint of this option is just over a third that of raw incineration as shown in Figure 4. The carbon impact of that technology train is not dissimilar to pre-MAD pasteurization. The reason why thermal hydrolysis further reduces carbon footprint with incineration is due to water, or more correctly, lack of it. Sensitivity analysis has shown that dry solids are the most influential parameter, even more so than volatile solids percentage, when burning biosolids (Spooren, 2007). When incinerated, biosolids need to have dry solids generally in excess of 35% in order to be autothermic. In Europe, in order to comply with the Waste Incineration Directive (WID) if the dry solids are below this figure, then pre-drying is required. This is achieved by diverting some of the steam generated by incineration away from energy production to an indirect dryer. This loses some of the benefits of energy generation, but also, more water in the system generates more condensate which requires costly (both in terms of economics and energy/carbon) treatment. Alternatively, natural gas must be burnt with the biosolids to enable their processing which would contribute significantly to carbon footprint.

With respect to composition, the carbon footprints of both raw and digested incineration are mainly similar with power and direct process emissions (of N_2O) accounting for approximately 30% each for both biosolids types. The one noticeable difference occurs during down-time, when it has been assumed that the biosolids are recycled to land. In this case, raw biosolids generate a higher quantity of GHGs when applied to land than do digested biosolids, therefore the impact of down-time for raw incineration is greater as shown in Figure 5k) 8% of total and 5l) just 2% of total.

Class B Processes

The contributions and benefits for carbon footprints with respect to Class B biosolids processes are shown in Figure 7), whilst Figure 8) shows the overall operating carbon footprint.

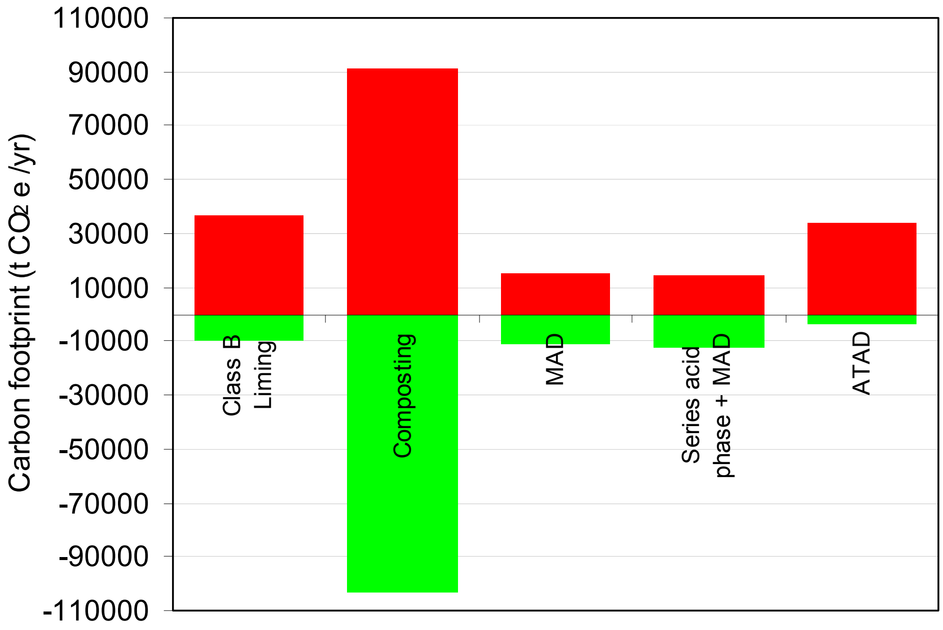

Figure 7. Carbon footprint contributions (🟥); and benefits (🟩) for Class B compliant processes. Note: operating carbon footprint calculated as sum of contributions minus benefits.

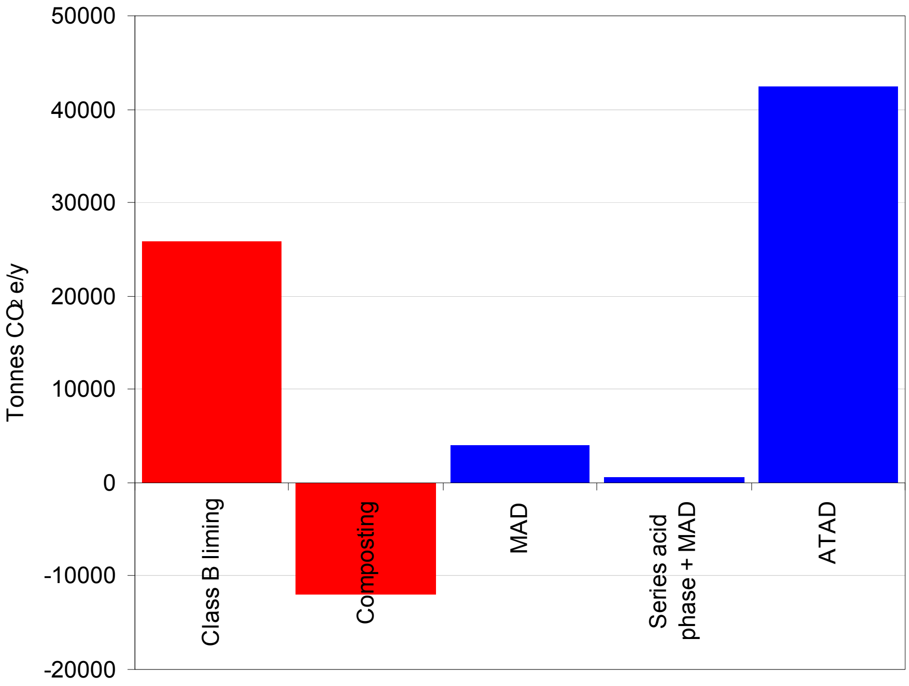

Figure 8. Operating carbon footprint for Class B compliant biosolids treatment processes. Key (🟥 processes without digestion; 🟦 processes with digestion)

The compositions of carbon footprints for Class B systems were very similar to their Class A equivalents. For example, the footprint for Class B ATAD can be described by Figure 5e). The noticeable exception to the rule was Class B liming. This is due to the reduced lime requirement when Class B status is required. Less lime addition reduces the carbon footprint contribution of the lime itself (from 37% for Class A to 17% for Class B). Factors such as dewatering; power and transport remain similarly influential whilst the difference is made up by the land emissions of GHGs (from 32% for Class A to 44% for Class B).

Generally Class B equivalents of Class A processes had lower carbon footprints as demonstrated in Figure 9.

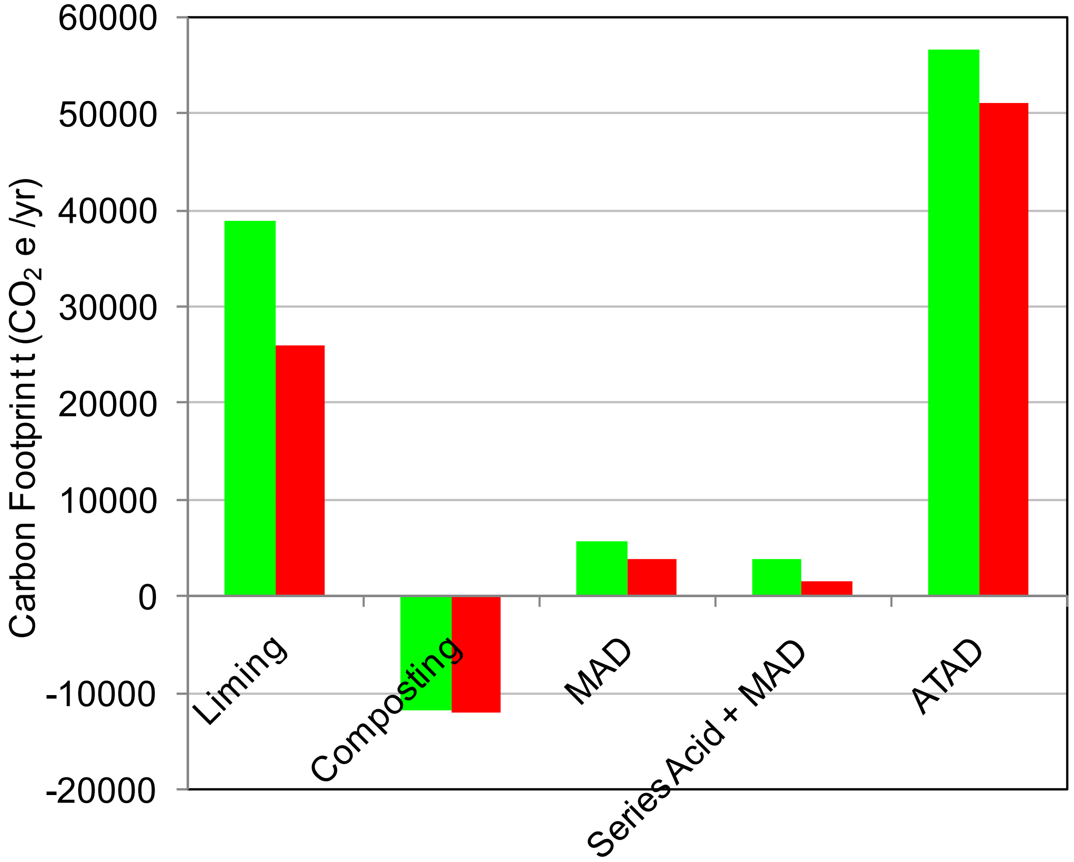

Figure 9. Influence of biosolids treatment requirement on operating carbon footprint. Key: Class A equivalent processes (🟩); and Class B equivalent processes (🟥). Note: Class A equivalent of MAD is assumed to be TAD-MAD.

For liming options, less lime is required for Class B systems as pasteurization temperatures are not required. The reduction in lime addition influences both, the carbon footprint from the lime itself, and transport emissions as there are less biosolids (plus lime) to transport. The overall impact is a reduction in carbon footprint of approximately one third. Interestingly, this is similar to the carbon footprint of the Class A lime plus supplemental heat process.

As for liming, the lack of necessity to reach a pasteurization temperature also reduces the carbon footprint of anaerobic digestion. When compared with thermophilic digestion, the calculated carbon footprint of MAD was 30% lower even though the Class A process was assumed to generate more energy. If energy generation in the TAD-MAD process was dropped to that expected of standard MAD, the carbon footprint would increase from approximately 5750 t CO_2e/yr (compared with MAD at 3990 CO_2e/yr) to over 13,300 CO_2e/yr – over three times higher than Class B MAD.

When looking at Class A (Enhanced Enzymic Hydrolysis) and B (Enzymic Hydrolysis) acid-phase in series followed by MAD, the opposite happens. The Class B equivalent of the technology generates more biogas energy than the Class A equivalent (Le, *et al*., 2007) and this has been demonstrated at full-scale where the Class A version destroyed 50% of the volatile solids in the biosolids (Werker, *et al*., 2007). A combination of lower energy demand (once again, as pasteurization is not required) and increased biogas production, makes the carbon footprint of the Class B configuration as much as 60% lower than its Class A version. After composting, series acid-phase digestion followed by MAD is the most favorable option with respect to carbon footprint to achieve Class B status.

The carbon footprint of Class B ATAD was approximately 10% lower than generated for the ATAD system to achieve Class A. Aeration requirements were identical for both, however the Class A configuration required supplementary heat to accommodate the combination of time/temperature used in this study. The Class B option had far higher mixing requirements due to the size of the plant needed to accommodate a 40 day HRT, but the increase in mixing power was lower than the additional heat needed for Class A. The study shows that depending on volatile solids destruction and retention time required there is little to differentiate between the carbon footprint generated from ATAD whether it is designed to achieve Class A or B status.

With regards composting, pathogen kill requirements had negligible impact on carbon footprint. This was mainly due to the influence of transport emissions. For both cases, regardless of process configuration, similar quantities of green waste would be required as compost dryness and carbon:nitrogen ratio requirements will be the same. For the composting itself, generation of GHGs will also be similar as an identical quantity of biosolids are being treated, however the period of time over which the emissions will be made will differ depending on whether the process was set up to provide Class A or B biosolids. Finally, the fertilizer displacement benefit will also be similar, due to the addition of identical quantities of green waste. However, it is possible to alter the carbon footprint of composting by selecting different green wastes, but this work was not done for this study.

Overall, the results of the model demonstrate that meeting more stringent legislative requirements for pathogens incurs additional carbon footprint.

DISCUSSION

There are a wide variety of biosolids processes which are capable of treating biosolids to comply with the pathogen reduction requirements as defined by US EPA 503 regulations. Choice of technology is complex and dependent on a number of project specific and economic influences. However, within Europe, long term sustainability and carbon impact are becoming more influential. Tax incentives and levies on the cost of carbon are also changing the economics in favor of more carbon-friendly processes. This work has shown that carbon footprints of processes to achieve the same level of pathogen destruction can vary widely. The salient conclusions are as follows:

- Class B equivalents of Class A processes had lower carbon emissions. This was primarily due to the energy required to meet pasteurization temperatures.

- Composting could have the lowest carbon impact if the benefits of fertilizer displacement can be argued for regardless of pathogen destruction level required. Otherwise, it will have the highest impact due to transport of green waste and biosolids.

- Processes with anaerobic digestion had far lower carbon footprints regardless if they were configured to achieve Class A or B status. Lower quantities of biosolids for downstream processing and transport, and the production of renewable energy to displace fossil-fuels were the main influences.

- For Class A processes, the lowest carbon footprints (after composting) are associated with advanced pretreatment prior to MAD. Thermal hydrolysis can further reduce carbon impact due to improved dewatering which in turn reduces down-stream processing requirements and transport when compared to similar processes.

- Digestion upstream of thermal processing – drying or incineration – can reduce carbon footprint by at least half compared to raw biosolids equivalents.

- When considering Class A liming systems, lime with supplemental heat systems are preferable due to lower lime requirements thus reducing the inherent carbon impact of the lime itself and reduced transport emissions.

- ATAD, either for Class A or B, had a high carbon impact due to large energy consumption for aeration and mixing. Alternative processes, if possible, are considered more sustainable in the long term.

- Pre-MAD liquid pasteurization had a far higher carbon footprint than similar process configurations to achieve Class A status with anaerobic digestion. This was due to the higher pasteurization temperature required.

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