Benchmarking the Sustainability of Sludge-Handling Systems in Small Wastewater Treatment Plants in Ontario

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ABSTRACT

This project quantitatively benchmarked all aspects of sludge handling in a cross-section of small wastewater treatment plants across Ontario. Using plant operational data and on-site measurements, a variety of sustainability metrics were evaluated: energy consumption, chemical use, biosolids disposition, biosolids quality, and greenhouse gas emissions. Overall electricity consumption ranged from 0.86 – 3.9 kWh per dry kg of raw sludge. The thermo-alkali hydrolysis and ATAD processes consumed the least (0.25 kWh/dry kg) and most (3.8 kWh/dry kg) amount of electricity for stabilization, respectively. Mechanical dewatering consumed minor amounts of electricity (2 – 5% of total sludge-handling draw), however, associated polymer dosages were found to be higher than literature values in some cases. The fuel requirements for disposition from such plants were up to 85% lower than facilities without dewatering. Biosolids contaminant (pathogen/metals) contents were observed to be substantially below NASM requirements. Four plants generated a Class A product, including one facility that generated it through a long-term storage approach (GeoTube™). Carbon emissions ranged from -119 to 299 kg CO_2 eq./dt. Six facilities exhibited net-negative emissions (-119 to -4 kg CO_2 eq./dt) with five of them employing process configurations that were relatively common province-wide. The benchmarking approach developed and information gathered will be of value to plant owners and operators who seek to better understand how their utility is performing relative to peers, identify areas of need and further investigation, and ultimately improve the long-term sustainability of their operations.

KEYWORDS

Biosolids, Sustainability, Residuals Management, Benchmarking, Sludge-Handling

INTRODUCTION

Conventional treatment of municipal wastewater involves the generation of semi-liquid sludge. The sludges are mostly water by weight (~98% prior to any processing), however, the solids portion contains several constituents of interest including organic material, nutrients, pathogens, and heavy metals (Metcalf & Eddy, 2013). Sludge generation represents a challenge from a plant operations stand-point, as it is continuously generated and must therefore be regularly processed and disposed of.

Moving forward, enhancing the long-term sustainability of wastewater treatment and associated sludge handling in small communities is of increasing importance to all stakeholders involved: owners, operators, and regulators. The practice of benchmarking is a strategy by which the

sustainability of sludge handling in wastewater treatment plants (WWTPs) may be improved. Such a practice can provide owners and operators with a tool to evaluate their plant's performance relative to others of similar capacity and scope of operation, and make informed decision-making based on the results.

Historically, much of the benchmarking of wastewater treatment operations has focused on a) broader, high-level metrics of overall WWTP process operations and performance (Vera et al., 2013; Yang et al., 2010), and b) large treatment facilities with advanced sludge processing (Bailey et al., 2014; Lindtner et al., 2008; Silva et al., 2016). Relatively little attention has been paid to small WWTPs (<10 MLD) that have limited capital, operating and human resources. Information gaps in the actual operation of such systems exist and the quality and disposition of biosolids from these systems is not well documented.

The objective of this study was to quantitatively benchmark the sustainability performance of a cross-section of sludge-handling systems in small WWTPs in Ontario. All analysis was based on actual plant data and on-site measurements to obtain the most accurate representation of existing performance. To achieve the objective, a systematic plant audit methodology was developed and implemented in ten WWTPs to evaluate a variety of sustainability metrics: energy consumption, chemical use, biosolids quality and disposition, and greenhouse gas emissions.

The information gathered will be of value to plant owners and operators that seek to enhance the sustainability of operations. The benchmarking approach developed can be applied to a broad range of small plants. Such an exercise can help small communities better understand how their utility is performing relative to peers of similar capacity and scope, identify areas of need and further investigation, and ultimately improve the long-term sustainability of their operations.

METHODOLOGY

For the purposes of the study, a "small" plant was defined as one with a design hydraulic capacity of less than 10,000 m^3/day that does not employ anaerobic digestion. Only mechanical treatment systems (liquid train and sludge stabilization) were considered for evaluation; lagoon systems were excluded as sludge generation at these facilities is sporadic. However, if a mechanical plant incorporated a lagoon as part of its non-stabilization sludge-handling process (*e.g.* for storage), it was still considered for selection.

To identify the facilities that met the initial screening criteria and would thus form the population of plants from which selections would be made, an Ontario Ministry of the Environment and Climate Change (MOECC) database containing basic plant information (location, hydraulic capacity, operator type, sludge treatment processes, disposition practice) of all facilities province-wide was analyzed. However, as the database was somewhat dated and incomplete in some areas, additional data was gathered on plants with hydraulic capacity greater than 1000 m^3/d to increase accuracy and completeness. The additional data gathering involved contacting municipalities directly and obtaining information from municipal websites. In total, 210 facilities met the initial screening criteria.

Plant Selection

From the 210 plants that met the initial screening criteria, ten were selected (Table 1) to capture a range of on-site sludge processing technologies (thickening, stabilization, dewatering), disposition practices (land application, landfill), operator type [public, private, Ontario Clean Water Agency (OCWA)], geographical locations (Southern, Eastern, Northern Ontario), and septage reception (present/not-present). Although most small facilities in the province are not currently using an "innovative" technology (*e.g.* thermo-alkali hydrolysis, GeoTube™, etc.) (Jin and Parker, 2017), it was desired to have such plants represented in the study to assess the extent to which newer technologies may impact the sustainability of operations, and provide baseline knowledge for other communities considering upgrades to or replacements of their existing process. Furthermore, although technologies such as centrifuge and rotary press dewatering are reasonably well-established among large WWTPs, they are not as common in small treatment facilities (Metcalf & Eddy, 2013) and thus would represent an innovation within the context of small plants. Facilities that employed these technologies were thus included in the current ten plant sample.

Table 1: Characteristics of Selected WWTPs

ID	Operator	Thickening	Stabilization Technology	Dewatering Technology	Holding / Storage	Odour Control	Disposition	Location	Septage Reception
A	OCWA		CAD		On-site lagoon		Agricultural	South	
B	Private	Gravity Thickener	CAD		Off-site lagoon		Agricultural	South	
C	Public		CAD	Centrifuge	Off-site storage	Biofilter	Agricultural	South	
D	OCWA		CAD	Rotary Press			Landfill	North	
E	Public		CAD	GeoTube™	GeoTube™		Agricultural	East	✓
F	Public		CAD		Aerated holding		Agricultural	South	
G	Public		CAD		Aerated holding		Agricultural	South	
H	Public	Gravity Thickener	Thermo-alkaline Hydrolysis	Centrifuge	Aerated holding (WAS), On-site storage	Biofilter	Agricultural	South	
I	OCWA		CAD	Off-Site Drying Bed			Landfill	North	
J	OCWA	Rotary Disc Thickener	ATAD	Rotary Press	On-site storage	Biofilter	Agricultural	East	✓

ATAD = authothermal thermophilic aerobic digestion
CAD = conventional aerobic digestion
OCWA = Ontario Clean Water Agency

Key Performance Indicator Selection

A variety of key performance indicators (KPIs) were established (Table 2) that could be broadly categorized into energy consumption, chemical use, biosolids disposition, biosolids quality, and greenhouse gas (GHG) emissions. The first four categories were selected to represent all the operational inputs and outputs of the systems studied, and to provide operational benchmarks for utilities seeking to quantify their individual plant performance relative to others of similar scale and scope of operation. The last category was selected as a means to cumulatively evaluate all previous categories on a common measure of environmental sustainability: the carbon footprint.

Table 2: Selected Key Performance Indicators

KPI	Category	Metric (Numerator) or Description						Normalizing Basis (Denominator)
1-6	Energy	kWh for thickening	kWh for stabilization	kWh for dewatering	kWh for pumping	kWh for odour control	kWh for aerated holding	dry kg of raw sludge produced
7	Energy	kWh for aerobic digestion						dry kg of VSS destruction
8	Energy	m³ of natural gas consumption						dry kg of raw sludge produced
9-10	Chemicals	kg of polymer used				kg of KOH used		dry tonne of raw sludge produced
11	Disposition	Weighted average round-trip distance to end-use destination (km)						N/A
12	Disposition	Liters of fuel consumed by haulage trucks						dry tonne of raw sludge produced
13	Quality	Mean TP content of hauled biosolids (g/kg)						N/A
14	Quality	Mean TN content of hauled biosolids (g/kg)						N/A
15	Quality	Mean K content of hauled biosolids (g/kg)						N/A
16	Quality	$\frac{\text{Mean metal concentration, dy mass basis}}{\text{NASM limit}}$ ratio of hauled biosolids						N/A
17	Quality	Mean log pathogen (*E. coli*) content of hauled biosolids [log (CFU/g)]						N/A
18	Quality	Biosolids product meets "NASM"/"Class B" requirements for land application: No = 0, Yes = 1						N/A
19	Quality	Biosolids product meets "Class A" requirements for land application: No = 0, Yes = 1						N/A
21-25	GHG	kg CO_2 eq. for electricity consumption	kg CO_2 eq. for natural gas consumption	kg CO_2 eq. for chemical consumption	kg CO_2 eq. for fuel consumption	kg CO_2 eq. for disposition		dry kg of raw sludge produced

Different energy sources (electricity, natural gas, transportation fuel) and chemicals have different emission debits associated with their respective production. Conversely, the land application of biosolids reduces chemical fertilizer requirements and thus provides the sludge-handling system with carbon credits. Taken collectively, a metric that converted all the inputs to a single net carbon footprint was used to evaluate the magnitude of environmental impact for each system.

Where appropriate, the KPIs were normalized on the basis of raw sludge (dry mass) produced, defined as the mass of sludge entering the sludge-handling process (from primary/secondary clarifiers) minus any mass quantities in return streams (*e.g.* digester decant and centrifuge centrate). The mass flows in the return streams were accounted for to ensure that facilities wasting large quantities of sludge did not receive a disproportionately favourable result if they were also returning high quantities back to the liquid stream, and thus had lower net sludge production than was apparent.

A BioWin™ model was generated for each plant to obtain a solids mass balance for the sludge-handling process, estimate net sludge production, and screen for problematic data. The modelling exercise involved initial configuration to reflect reported operating conditions [influent/effluent characteristics, flows (influent, waste/return sludge), chemical addition(s)] based on three years of historical operational data (2014 – 2016, if available). Unknown return streams (*e.g.* digester decant, dewatering centrate) were then adjusted such that the predicted aeration basin mixed-liquor suspended solids (MLSS) matched the reported values and predicted biosolids quantities matched the reported amounts (if available).

KPI Category 1: Energy Consumption

For all but one of the plants studied, electricity was the only form of energy consumed. Several of the energy KPIs thus involved normalized electricity consumption (kWh per dry kg of raw sludge) for the sludge-handling process. Individual electricity consumption KPIs for each stage of the sludge treatment process (thickening, stabilization, dewatering, holding), odour control, and pumping were selected to ensure information was obtained for each individual unit process. Additionally, recognizing that nine of the ten plants practiced some form of aerobic digestion, an additional indicator was selected to relate digester electricity consumption to the quantity of volatile solids reduction. The metric was selected to obtain a measure of the energy efficiency of the process. The quantity of solids destroyed was estimated from the BioWin™ model of each plant.

To determine the power draw of the various processing equipment (blowers, pumps, dewatering units, etc.), spot measurements were collected on-site using a Fluke™ 1735 power logger. Power draw was assumed to be constant over time since no major pieces of equipment incorporated variable frequency drives. Centrifuge back drives were the only exception, however, draw for these motors were found to only represent 5% of the total draw for the dewatering unit. The variation in draw was therefore assumed to be negligible. Electricity consumption (kWh) was estimated by multiplying daily equipment run-times (obtained from plant records and discussions with plant operators) with measured power readings (kW).

In most cases, the facilities employed dedicated blowers for aerobic digesters and holding tanks. Hence, power draw was directly allocated to the sludge handling process of interest from the measurements taken on-site. However, there were some instances where the same blower supplied air to both digesters and aeration basins (plants B, F, I, and J), or to both the digester and aerated holding tank (plant G). In the cases of plants B, G, and J, information on the air flow to each vessel was obtained to determine the percentage of air (and in turn, the proportion of electricity) supplied to the processes of interest.

Air flow information was not available for plant F. Hence, diffuser information and dissolved oxygen (DO) concentrations were employed in the BioWin™ model to estimate air flows and the corresponding allocation of power draw. Neither flow information nor diffuser information were available for plant I. Therefore, the proportioning was estimated based on the percentage of volume present in the aerobic digesters and the extended aeration basin. The need for these estimates introduced some uncertainty into the estimated KPIs for these five plants.

An additional energy KPI reflected the use of natural gas at plant H. It was calculated by subtracting the reported baseline usage (for plant-wide heating) from the total draw reported during stabilization operation, and dividing the difference by the dry mass flow of sludge processed.

KPI Category 2: Chemical Usage

While several of the facilities only used chemicals in the liquid train (for phosphorus removal), those that practiced dewatering or mechanical thickening used polymer to enhance the liquid-solid separation process. In addition, one of the facilities (plant H) used potassium-hydroxide

(KOH) for pH control and to boost the potassium content of the biosolids product. Two KPIs were selected to reflect these inputs. Chemical usage information was obtained from plant records and/or conversations with plant operators. However, the specific form in which the information was available was not consistent across all plants. Specific usage quantities were calculated using reported chemical purchase records (plant C), barrels/volumes consumed per month (plants D and J), dosing rates (plants E), and flow rates (plant H).

KPI Category 3: Biosolids Disposition

Separate indicators that employed the average distance that the biosolids travelled to their destination and the amount of fuel consumed (normalized to dry mass of solids processed) were created. The latter indicator was chosen to account for the variety in capacity and fuel economy of the trucks in use. Liquid biosolids are typically transported in large tanker trucks with capacities of approximately 40 m^3 per truck, while dewatered cake is often hauled in small-to-medium sized dump trucks that have smaller capacities and lower fuel requirements.

Biosolids disposition information [quantities and farm/landfill address(es)] was obtained from haulage reports and Google Maps™ was employed to determine the shortest driving distance from the WWTP to each destination. To calculate the normalized fuel consumption of each operation, the distance value was used in conjunction with truck fuel economy information obtained from the truck owner.

KPI Category 4: Biosolids Quality

Biosolids contain nutrients [phosphorus (P), nitrogen (N), potassium (K)] that are beneficial for agricultural crop growth, but also contain heavy metals and pathogens that can be harmful at high concentrations. Limits for the latter two measures have been established in the Nutrient Management Act for Non-Agricultural Source Material (NASM) application (O. Reg. 267/03 – Schedule 5 and 6, CM2 and CP2). Further distinctions regarding pathogen content are stipulated under the US EPA regulatory framework (US EPA, 1993). Under EPA guidelines, a "Class A" product must contain less than 1000 MPN/g of *E. coli*, while a "Class B" product must have less than $2x10^6$ CFU/g (US EPA, 1993). The latter value is consistent with the NASM requirement in Ontario (O. Reg. 267/03 – Schedule 5, CM2). In Canada, if a biosolids product meets thresholds for pathogen content, it can qualify as a CFIA-certified fertilizer under the Fertilizers Act (R.S.C., 1985, c. F-10) and associated Fertilizers Regulations (C.R.C., c. 666).

KPIs for mean nutrient and geomean *E. coli* concentrations were selected to identify the range of beneficial value (nutrients) and proximity to the NASM pathogen limit (*E. coli*) of the hauled biosolids. In addition, a KPI relating the highest ratio of mean metal concentration to its respective NASM limit was selected to determine whether any metals were at risk of exceeding regulatory thresholds. In addition, binary indicators were included to represent whether the biosolids were a) meeting NASM requirements for land application, and b) meeting requirements for classification as a Class A product. Quality data was obtained from plant records for eight of the ten plants practicing land application of biosolids. For the two plants landfilling their biosolids, such information was not available. To determine the quality characteristics for the latter two plants, a sampling program was implemented to characterize the biosolids product leaving the plant (cake and liquid for plants D and I, respectively). The sampling program

involved measuring all parameters of interest (nutrients, *E. coli*, metals) for four months on a bi-weekly basis. All samples were collected by plant operators and sent to MOECC accredited labs for analysis.

KPI Category 5: Greenhouse Gas (GHG) Emissions

CO_2 emissions were calculated for each facility on the basis of emission factors that were obtained from the literature for each input and output. Where possible, emission factors specific to Ontario (electricity, natural gas production) or Canada (transportation fuel) were employed. In other cases, literature values for chemical production (polymer, KOH) and chemical fertilizer production (N, P, K) were utilized. The latter factors were used to determine the carbon off-sets gained by using biosolids as a fertilizer through the avoidance of chemical fertilizer production for each nutrient.

RESULTS

The benchmarking results associated with each system input and output were evaluated. Where possible, uncertainties in the estimated values (expressed as the standard deviation as a percentage of the mean for each KPI) were calculated from the raw data set. Among the KPIs that involved raw sludge production, the variability in production represented the largest source of uncertainty. The standard deviations associated with raw sludge production were found to range from 11 – 34% of the mean values. The variability was attributed to differences in biological activity (biomass growth), influent fixed suspended solids, and chemical sludge production (if chemical addition for P-removal is practiced). No individual observed value was noteworthy, however there were two clear "tiers" of values observed: a lower tier wherein uncertainties were between 11-15% (four plants), and a higher tier where values ranged between 27-34% (six plants). There was also unquantifiable uncertainty associated with a) parameter estimates provided by operators (*e.g.* polymer use), and b) partitioning of electricity draw when one blower supplied air to both liquid stream aeration basins and digesters. These qualitative uncertainties were described in the methodology section. Where necessary, the implications of such uncertainties (quantitative and qualitative) on the extent to which conclusions may be drawn are discussed in subsequent results sections.

Energy KPI Results

Energy inputs represent a portion of a treatment facility's total operational costs and the GHG emissions associated with their production represent an environmental impact. As such, reductions in this area without compromising plant performance can potentially improve economic and environmental sustainability. The following discussion details the KPIs related to electricity and natural gas consumption.
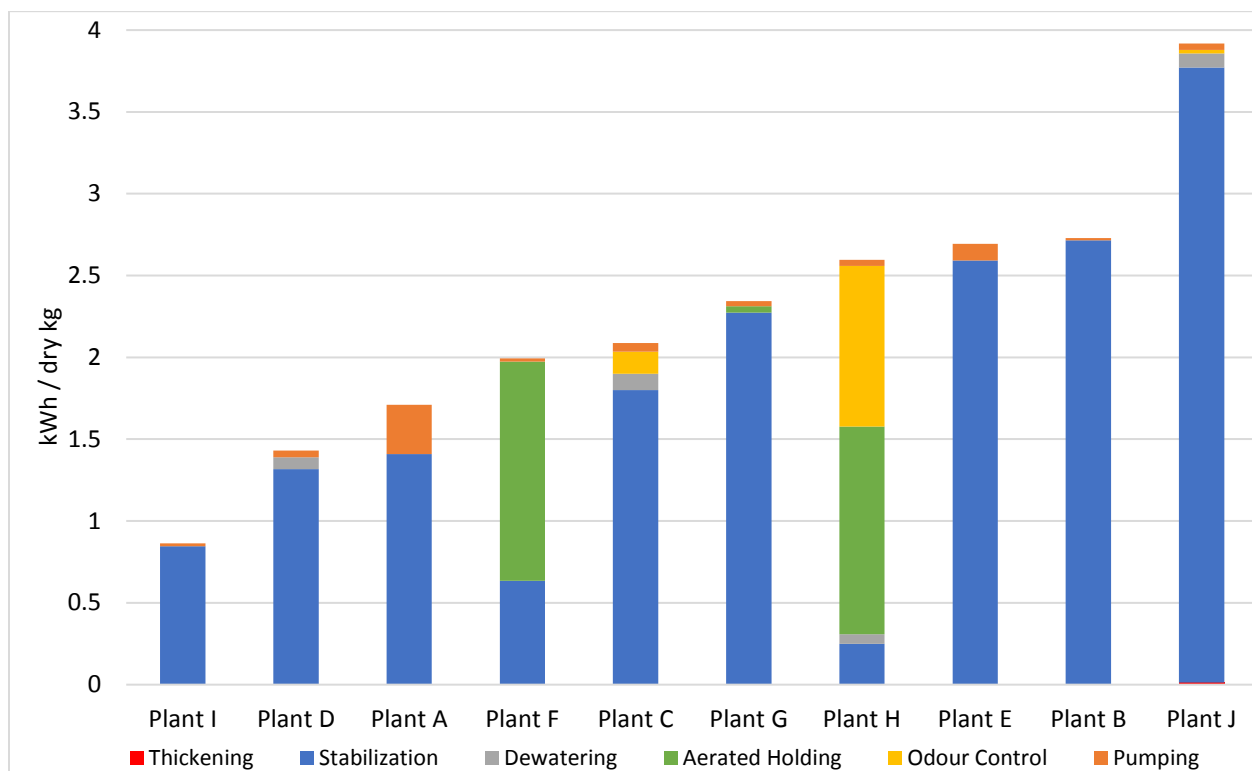
Electricity Consumption – Overall

All ten plants within the study consumed electricity as part of the sludge-treatment process. Total electricity consumption with associated uncertainty for each facility is presented in Figure 1, while the contributions of individual processes to total consumption is shown in Figure 2. As shown in Figure 1, total consumption ranged from 0.86 – 3.9 kWh/dry kg of raw sludge among

all plants studied. The 25th, 50th and 75th percentile values corresponded to 1.8, 2.2, and 2.7 kWh/dry kg, respectively.
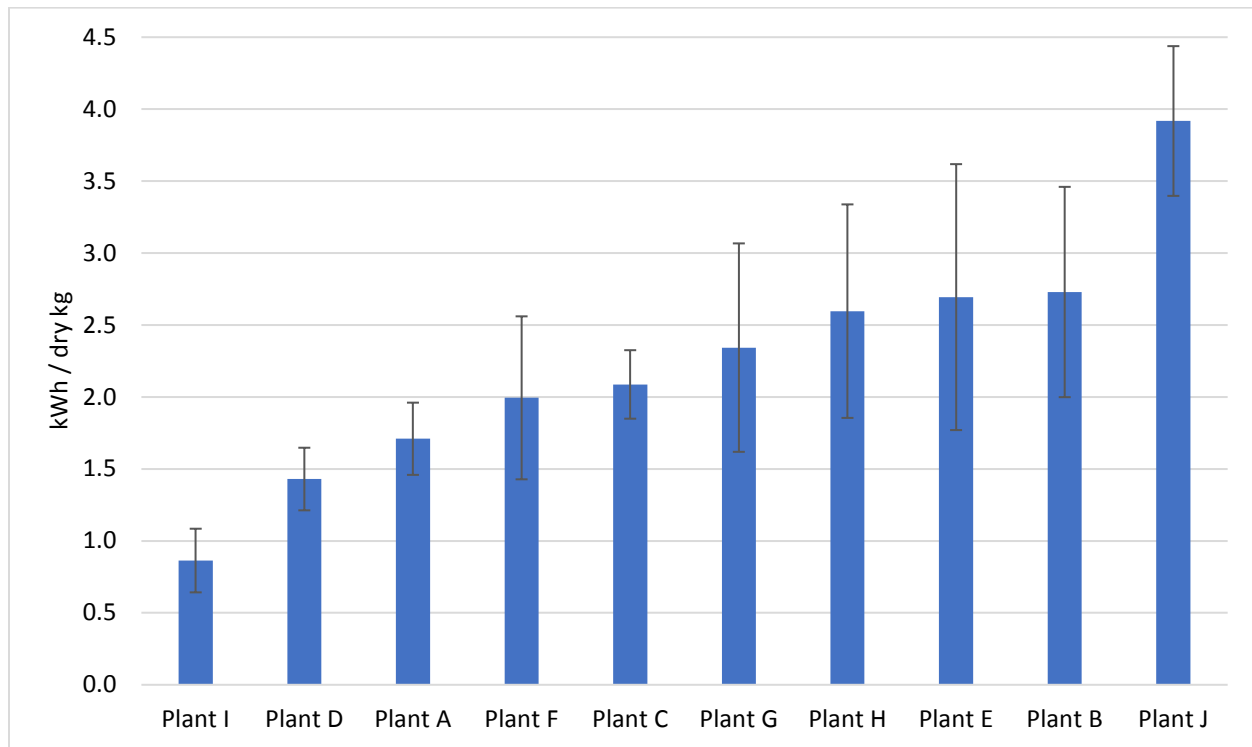
Figure 1: Total electricity consumption per dry mass of raw sludge produced

Figure 2: Detailed electricity consumption per dry mass of raw sludge produced

When incorporating quantitative uncertainty into the analysis, it can be seen from Figure 1 that there was overlap in the uncertainty bars between all plants inclusive of the second (plant B) and eighth (plant A) highest consumers. The considerable overlap indicated that total electricity consumption was similar for a number of plants in the study. Notably, the lowest consumer (plant I) exhibited consumption that was statistically different than the next closest consumer (plant D). However, as discussed in the methodology, there was unquantified uncertainty in plant I estimates as it did not employ a dedicated digester blower. Hence, the low KPI for plant I may not be a feasible goal for plants seeking to reduce electricity consumption.

When the type of sludge-handling technologies employed was considered, the facilities that did not practice conventional aerobic digestion exhibited the highest (ATAD) and fourth-highest (thermo-alkali hydrolysis) electricity consumption, respectively. The former facility consumed 44% more energy per unit of raw sludge mass than the next highest consumer, indicating that the ATAD technology was substantially more energy-intensive than the conventional aerobic digestion processes within the sample. Among the eight plants practicing conventional aerobic digestion, total electricity consumption ranged from 0.86 – 2.7 kWh/dry kg. For such plants, the 25th, 50th and 75th percentile corresponded to 1.6, 2.0, and 2.4 kWh/dry kg, respectively.

Two facilities (E and J) that practiced septage reception were evaluated to assess the impacts of this practice on KPI values. Septage is a partially stabilized material that directly contributes to sludge production via fixed suspended solids loading (Metcalf & Eddy, 2013). It was expected that plant would exhibit higher sludge production and aeration basin MLSS than predicted (modelled) values as the modeling did not account for this input. Plant E exhibited a lower MLSS value than the simulated concentrations. The facility also exhibited the highest uncertainty in raw sludge production (34%) among all plants studied. In addition, plant E was operated at a higher solids retention time (SRT) than all other facilities studied. Collectively, these factors contributed to the difficulty in ascertaining the impact of septage reception on the energy consumption at the plant. Plant J was an ATAD facility that required substantially greater energy input as a result of the chosen stabilization process, which resulted in difficulty extracting the energy consumption due to septage reception. It did generate similar quantities of sludge to the predicted value, which indicated that increased solids loading from septage was sufficiently represented by the BioWin™ model. Ultimately, no conclusions could be drawn regarding the impact of septage reception on sludge-handling energy requirements because both case studies exhibited additional factors that could not be de-lineated from the reception of this material.

Electricity Consumption – Stabilization

As shown in Figure 2, the electricity allocated to stabilization represented the highest fraction of sludge-handling electricity consumed for all but two of the plants studied (F and H). With the exception of plants F and H, at least 82% of the electricity consumption in all of the facilities was used for stabilization. The high proportion of electricity utilized for stabilization suggests that this process should generally be an area of interest for plant owners and operators that seek to selectively target high usage unit processes within their overall treatment system. A reduction of the electricity required for stabilization would have a greater impact on total usage reduction than the same percent reduction achieved within other processes (*e.g.* dewatering).

Normalized power consumption values for the stabilization processes alone (with associated uncertainties) are shown in Figure 3. Among all the plants studied, electricity consumption for stabilization ranged from 0.25 to 3.8 kWh/dry kg. The 25th, 50th and 75th percentile values corresponded to 1.0, 1.6, and 2.5 kWh/dry kg, respectively. However, the results differed from the overall consumption results in that any given plant's uncertainty bar in Figure 3 generally overlapped with fewer other plants than those in the previous (overall) analysis. The most overlap any single plant exhibited in Figure 3 was four facilities, whereas some facilities exhibited as many as six overlapping values in Figure 1. The observation indicates that the ranking of consumers with respect to stabilization draw was more defined than that of the overall ranking. Furthermore, given that all the plants employed the same technology, the over two-fold increase in consumption between the 25th and 75th percentile indicates that opportunities for process optimization in some of the higher consumers may exist.
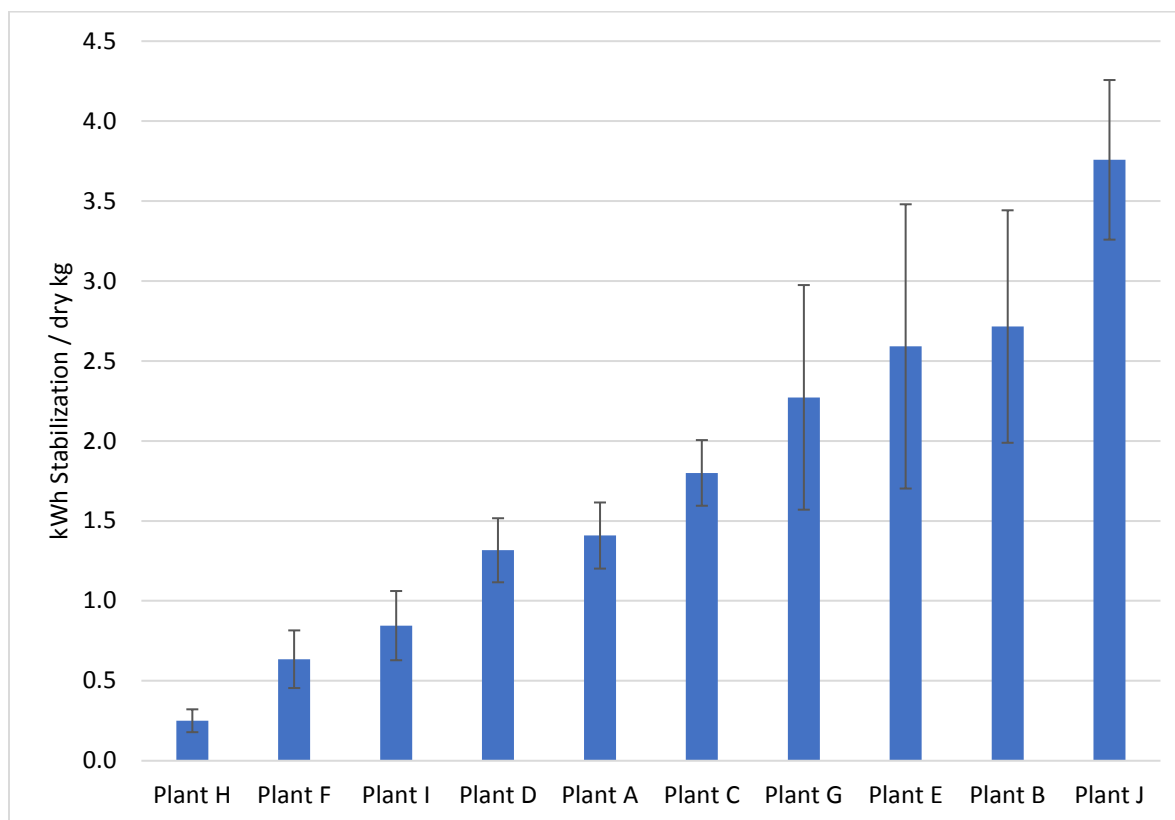

Figure 3: Stabilization electricity consumption per dry mass of raw sludge produced

The maximum consumption associated with stabilization corresponded to the facility that employed ATAD, while the minimum value corresponded to the plant that employed thermo-alkali hydrolysis. The latter plant's uncertainty bars did not overlap with any other facility, which indicated that it was also the best performer when quantitative uncertainty was incorporated into the analysis. The observation that thermo-alkali hydrolysis was the lowest consumer for stabilization is noteworthy when considering the technology's application in other facilities. If the thermo-alkali hydrolysis process were to be implemented at a plant that did not require aerated holding or odour control (the two largest power consumers for sludge handling operations at plant H), the potential for electricity savings could be substantial relative to the conventional aerobic digestion process.

Given that conventional aerobic digestion is the most common stabilization technology employed at small WWTPs in Ontario (Jin and Parker, 2017), the percentile benchmarks for plants employing this technology were evaluated separately. Among such plants, digester electricity consumption ranged from 0.63 – 2.7 kWh/dry kg, while the 25th, 50th, and 75th percentile corresponded to 1.2, 1.6, and 2.4 kWh/dry kg, respectively.

In addition to the analysis based on stabilization energy consumption normalized by raw sludge production, the power consumption of facilities that practiced aerobic digestion was evaluated on the basis of VSS destruction achieved (Figure 4). This measure provided an indicator of the energy efficiency of the digestion process, given that VSS destruction is one of the primary functions of an aerobic digester (Metcalf & Eddy, 2013). As shown in Figure 4, the ATAD plant exhibited the highest energy consumption on this basis, consuming 63 kWh per dry kg of VSS destroyed. Among the eight facilities employing conventional aerobic digestion, consumption ranged from 4.9 – 56 kWh/dry kg VSS. The 25th, 50th, and 75th percentile corresponded to 7.1, 8.7, and 19 kWh/dry kg VSS, respectively.
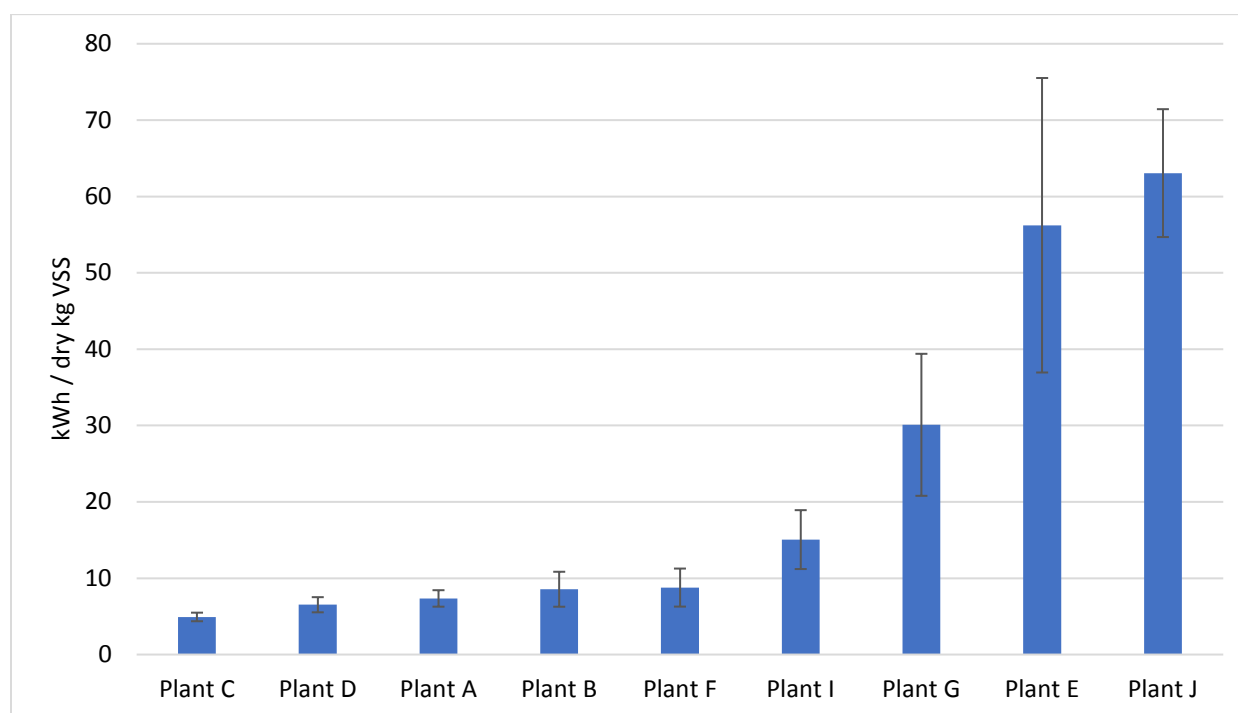
Figure 4: Digester electricity consumption per dry mass of VSS destruction

The minimum value (4.9 kWh/dry kg VSS) was statistically lower than any other value observed (no uncertainty bar overlap with other facilities), while the four next highest consumers (6.5 – 8.8 kWh/dry kg VSS) exhibited statistically equivalent consumption. Thus, the five lowest consumers could collectively serve as benchmarks for other utilities seeking to determine their performance relative to peers of similar scope and operation. The low consumption in plant C may have been due to the type of sludge being digested. The facility generated a mix of primary and secondary sludge, the former of which is more readily biodegradable than secondary sludge (Metcalf & Eddy, 2013). The resulting mixture thus generally requires less air to achieve a given quantity of VSS destruction than pure secondary sludges (Metcalf & Eddy, 2013). One other facility generated primary sludge (plant B), and exhibited the second lowest specific energy

consumption (uncertainty bar overlap with three facilities), despite being the second highest overall consumer (Figure 1).

Plants G and E exhibited over three- and six- fold greater energy consumption than the median value, respectively. Both facilities exhibited higher specific consumption than the next highest consumer (plant I) even when considering the quantitative uncertainty associated with the raw sludge production (neither plant's uncertainty bar overlapped with plant I). To provide insight into why each facility exhibited notably higher consumption, the hydraulic residence time (HRT) of each facility was examined and found to have HRTs of 48 and 58 days respectively. Both values were substantially higher than the MOECC (2008) design guideline of 15 days HRT, which suggests that both digesters are a) oversized based on the current loading to the digester, and/or b) have air requirements for mixing that exceed the air supply requirements for VSS destruction. Essentially, both units were effectively operating as aerated holding tanks in addition to their function as aerobic digesters.

The energy efficiency evaluation revealed that some of the facilities that performed best when evaluated on a raw sludge production basis were among the worst performers when evaluated on the basis of VSS destruction achieved, and vice versa. Among plants that practiced conventional aerobic digestion, plant I exhibited the second lowest energy consumption when normalized by raw sludge production, but the third highest consumption when normalized by VSS destruction achieved. Plant B exhibited the highest consumption under the former basis, but a value less than the median when evaluated on the latter basis. While plants G and E were among the highest consumers when normalized by raw sludge production, the extent to which they were the highest consumers when normalized by VSS destruction was substantially greater than when evaluated under the previous basis. As previously discussed, the digesters for both plants G and E were likely oversized based on the current VSS loading, which contributed to the high specific consumption observed.

Overall, the examination of energy efficiency in stabilization provided insight into areas where improvement might be possible, and highlighted possible deficiencies that would not have been identified had energy consumption only been calculated on a raw sludge production basis. Given the broad range of values observed, opportunities for improvement from an energy efficiency basis may exist in several of the facilities studied.

Electricity Consumption – Dewatering

Dewatering is employed to convert liquid biosolids into a cake. Of the five plants that practiced dewatering, four employed a mechanical process that required electricity as part of its operation while the fifth plant employed a passive process (GeoTube™). Among such plants, the normalized power draw for dewatering ranged from 0.058 – 0.10 kWh/dry kg (Figure 5). The minimum and maximum values corresponded to centrifuge processes and did not exhibit overlap of the uncertainty bars (indicating distinctly higher and lower consumption between the two samples). The two rotary presses consumed between 0.073 – 0.087 kWh/dry kg, however, overlap between the uncertainty bars indicated that there was no statistical difference between the two values. Notably, the percentage of total sludge-handling electricity consumed by the dewatering processes ranged from 2% (plants H and J) to 5% (plants C and D). The low

percentages of total sludge-handling power draw indicated that the additional energy required to convert liquid sludges into cake via mechanical dewatering was relatively minor.
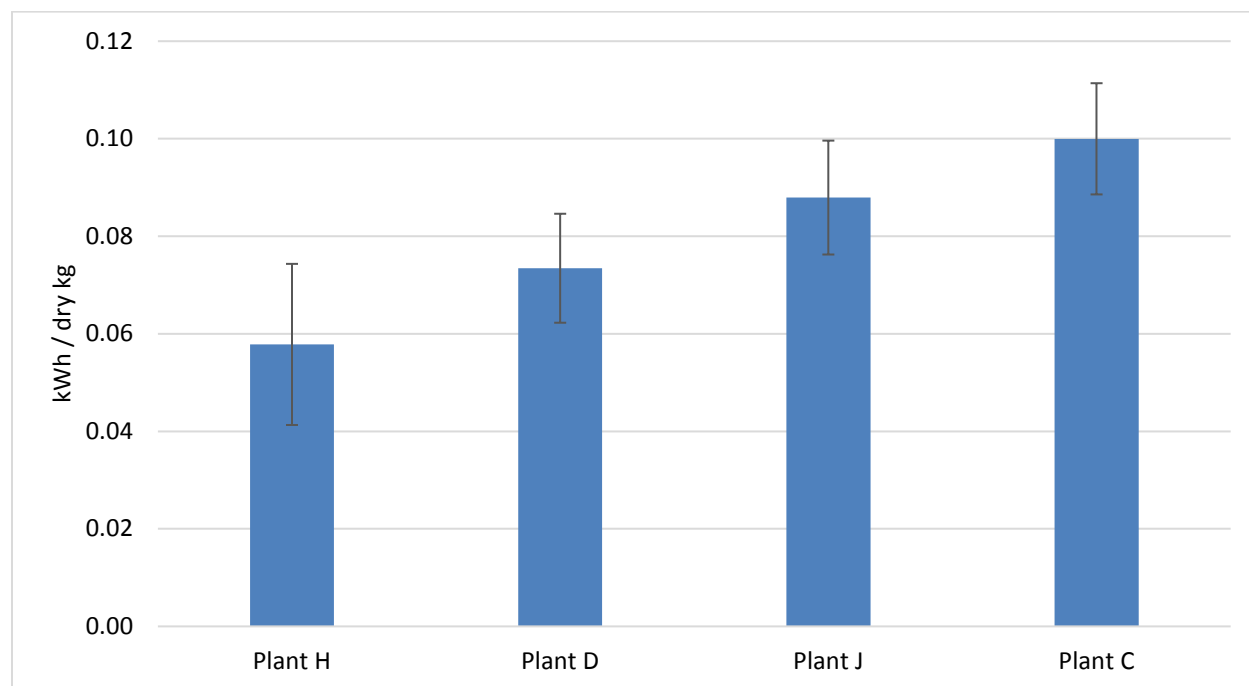
Figure 5: Mechanical dewatering electricity consumption per dry mass of raw sludge produced

Electricity Consumption – Pumping

For all but one of the plants studied, pumping represented a minor percentage of the total sludge handling draw (1 – 4 %). Among all plants studied, the 25th, 50th, and 75th percentile of pumping electricity consumption corresponded to 0.023, 0.039, and 0.050 kWh/dry kg, respectively. Plant A represented an extreme value in this regard where pumping represented 18% of the total draw. Its normalized consumption was 0.30 kWh/dry kg, which was a three-fold greater consumption than the next highest consumer (0.10 kWh/dry kg). The identification of the cause for the high use was beyond the scope of the study, but a possible explanation involved the solids content of the feed sludge. Specifically, the wasted secondary sludge was dilute (~ 0.5% TSS), which may have resulted in increased pumping requirements to waste the desired mass of sludge.

Electricity Consumption – Aerated Holding

Among the three plants that employed aerated holding of sludge, plants F and H exhibited similar consumption for the process (1.3 kWh/dry kg), while plant G exhibited a substantially lower value (0.040 kWh/dry kg). Both plants F and H employed a dedicated blower for their holding tank, while plant G utilized a portion of the air provided by its digester blower to aerate the holding tank (the reported air flow to each unit was used to determine its corresponding allocation of electricity consumption). The discrepancy in values may be explained by the observation that plant G employed an extended HRT in its digester (48 days), which indicated that it was effectively employing its digester as a holding tank.

Electricity Consumption – Odour Control

There was a broad range of electricity consumption values associated with odour control: plants J, C, and H consumed 0.021, 0.13, and 1.0 kWh/dry kg, respectively. The underlying cause of the wide range was beyond the scope of the study, however site-specific considerations likely influenced the quantity of electricity required to eliminate odours. All three facilities employed biofilters to remove odours. However, only the facility that practiced thermo-alkali hydrolysis (plant H) employed a highly engineered system and this corresponded to the highest normalized power draw. Since plant H did not practice aerobic digestion (which aids in odour removal), its odour control system would need to remove all the odours generated by the sludge, which likely increased the energy input requirements. One would have expected the ATAD system to require higher electricity requirements since ATADs have historically been associated with considerable odour emissions (Metcalf & Eddy, 2013). However, the sample facility employed a relatively new "second-generation" ATAD process which is substantially less odourous than early "first generation" systems (Metcalf & Eddy, 2013). One could also hypothesize that the ATAD process required lower volumes of air for treatment than the thermo-alkali hydrolysis process, thereby reducing the electricity requirements for odour control.

Natural Gas Consumption

In the current study, one facility consumed natural gas as part of its sludge treatment process (plant H). The observed consumption was 0.037 m^3 natural gas/dry kg. The lack of natural gas usage within the study sample suggests that it is not a common form of energy employed at small WWTPs for the purposes of sludge processing. Indeed, anecdotal conversations with owners and operators revealed that natural gas was typically only used to heat office buildings, if it was used at all.

Chemical Usage KPI Results

Chemical use is necessary to achieve the goals of some treatment processes. For all the thickening and dewatering processes within the study, polymers were used to enhance the liquid-solid separation process. In addition, KOH was employed as part of the stabilization process in one instance. The purchase of chemicals represents an operational cost for plant owners and the GHG emissions associated with their production represent an environmental burden.

For the plants within the study that employed thickening and/or dewatering processes, normalized polymer usage and the corresponding biosolids cake total solids (TS) content are shown in Figure 6. Among the selected plants, one plant (J) employed a thickening technology (rotary disc thickener), one plant (E) employed a passive dewatering technology (GeoTube™), and four plants (C, D, J, H) employed mechanical dewatering processes. Plants C and H employed centrifuges, while plants D and J employed rotary presses.

The rotary disc thickener consumed 12 kg of polymer per dry tonne (dt) of raw sludge and generated a 4.5% TS sludge product. The GeoTube™ consumed 9 kg polymer/dt and generated a 9.2% TS product. Polymer usage for rotary press operation ranged from 20 – 28 kg polymer/dt (no uncertainty bar overlap) and generated a biosolids cake ranging from 16.9 – 18.4% TS (uncertainty bar overlap). Polymer usage for centrifuge operation ranged from 8 – 24

kg polymer/dt (no uncertainty bar overlap) and generated a biosolids cake ranging from 17.2 – 22.5% TS (no uncertainty bar overlap). Notably, the lower centrifuge chemical usage value corresponded to the higher TS content. The lower dosage was employed at the plant that generated a mixed primary/secondary sludge, which typically exhibits higher dewaterability than pure secondary sludges (Metcalf & Eddy, 2013).
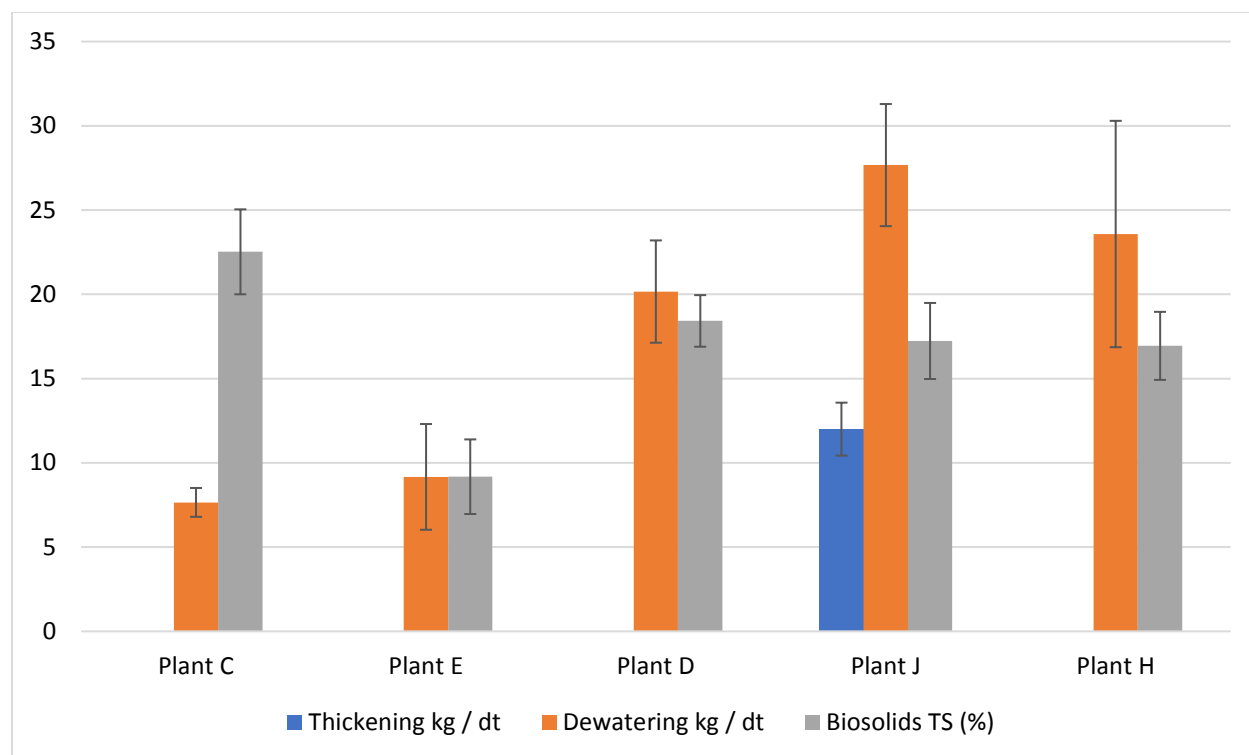
Figure 6: Chemical usage per dry mass of raw sludge produced and biosolids TS content

Among the two rotary disc thickeners evaluated, the polymer usage and solids content extended beyond the range of values reported in the literature. Rotary presses employed for dewatering aerobically digested waste activated sludges have been reported to consume a maximum of 17.5 kg polymer/dt and achieve 28 – 45% solids (Metcalf & Eddy, 2013). For centrifuge use, literature indicates that 10 – 15 kg polymer/dt can be employed to achieve between 18 – 25% solids (Metcalf & Eddy, 2013). The observed discrepancies between observed and literature values may indicate that excess polymer was being dosed in some cases, or that the polymers employed were less effective as coagulating/flocculating agents than those reported in literature.

In addition to polymer usage for dewatering, plant H utilized KOH as part of its stabilization process. The observed usage was 19 kg KOH/dt. The relative impact of the KOH and polymer use at this plant on GHG emissions will be detailed subsequently.

Overall, there was a broad range of chemical use employed at the facilities within the sample. Rotary thickening and GeoTube™ dewatering generally consumed the least with 12 and 9 kg/dt, respectively. Rotary press and centrifuge dewatering operations consumed 20 – 28 kg/dt and 8 – 24 kg/dt, respectively. Three of the mechanical dewatering usage values were higher than those reported in literature. The observation suggests that the polymers may have been overdosed in the sample cases, or that the chemicals used were less effective as coagulants/flocculants than

those reported in literature. One facility utilized KOH as part of its stabilization process and consumed 19 kg KOH/dt.

Disposition KPI Results

The disposal of biosolids represents an operational cost for wastewater treatment plant owners, and the carbon emissions associated with trucking fuel consumption represent an environmental burden. Disposition KPIs involving the average distance that biosolids were hauled and the associated normalized fuel consumption were evaluated (Figure 7). For the former calculation, uncertainty was assumed to be constant (0.5 km) since Google Maps™ was employed to obtain the exact address of each farm/landfill and uncertainty was therefore only associated with variation in distance travelled within a given farm. Uncertainty for the fuel consumption was directly a result of the uncertainty in estimated raw sludge production for each plant.
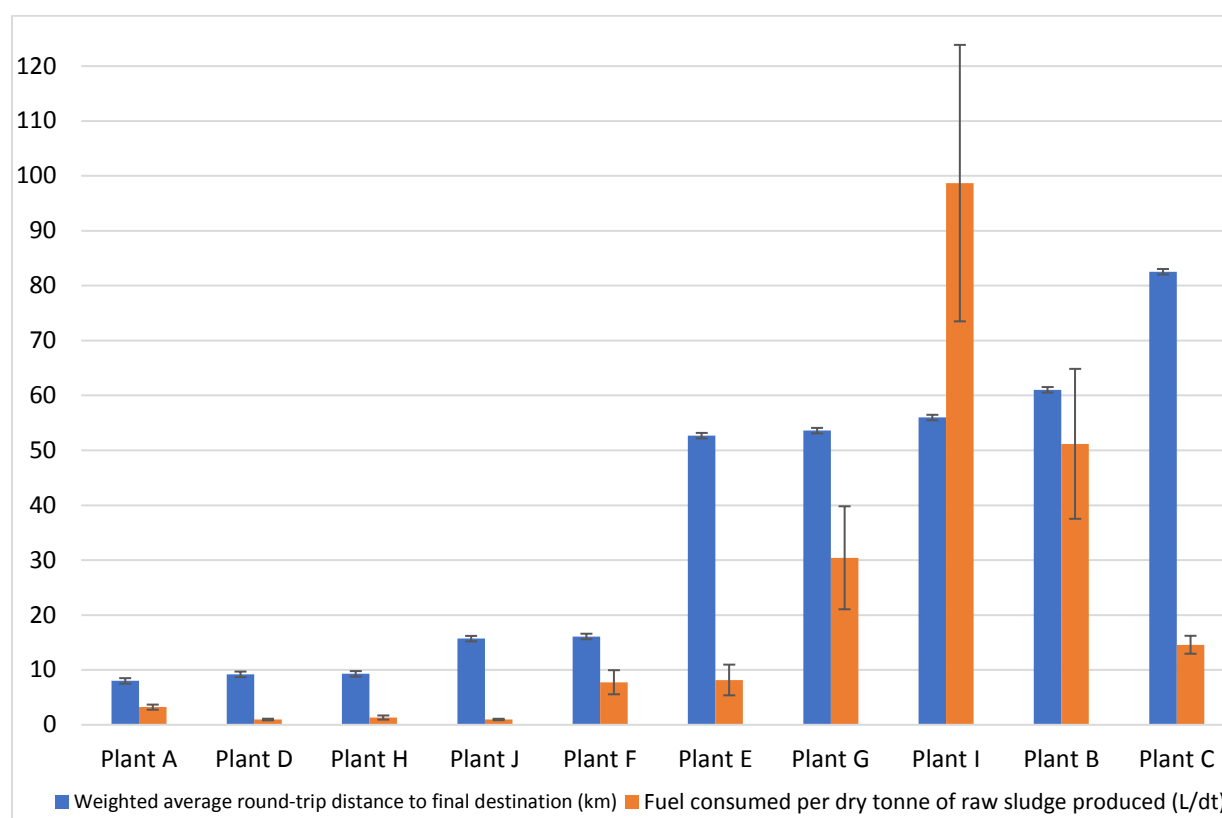
Figure 7: Disposition KPI results

From Figure 7 it can be seen that the average round-trip trucking distance ranged from 8 – 83 km. There was a substantial difference in trucking distance requirements between the lower and upper five facilities studied. Half of the plants required less than 16 km of round-trip trucking to dispose of their biosolids, while the other half required more than 52 km. The three highest distances corresponded to plants (C, B, I) that did not have on-site storage, which necessitated additional trucking. In each case, transport involved trucking the biosolids from the WWTP to an off-site location (storage building, lagoon, drying bed) and trucking from the off-site location to the final destination (farm, landfill) at a later date.

As shown in Figure 7, fuel consumption normalized by raw sludge production ranged from 1 – 99 litres per dry tonne of raw sludge produced. The broad range observed was influenced by multiple factors: the distance required for trucking, the capacity and fuel economy of the trucks employed for transportation, and the presence (or lack thereof) of on-site dewatering. Notable observations were obtained when comparing facilities with similar trucking distances, but different dewatering practices. Among the three plants with the highest trucking distances, only plant C employed on-site dewatering (Plant I employed an off-site drying bed). As a result, plant C consumed 85% and 72% less fuel than plants I and B, respectively. Plants E and G required similar trucking distances (53 and 54 km, respectively), but the former facility consumed 75% less fuel as a result of on-site dewatering (GeoTube™). Similar observations were made for plants J and F, which exhibited similar trucking distances and a substantial difference in fuel consumption for the facility that employed dewatering (plant J).

A broad range of trucking distance requirements and normalized fuel consumption quantities were observed among the facilities studied. The difference between trucking distance and fuel consumption was largest when comparing facilities that employed chemically enhanced dewatering against those that did not. The results indicate that the implementation of dewatering processes to reduce trucking requirements is a consideration worthy of investigation. From an environmental sustainability standpoint, the reduced fuel consumption represents a savings in the carbon footprint associated with trucking. However, the manufacture of chemicals represents a source of GHG emissions.

Biosolids Quality KPI Results

The quality (pathogens, nutrients, and metals content) of the biosolids product is an important consideration when considering the end-use of the biosolids product. For facilities that wish to land apply their biosolids for agricultural use in Ontario, both pathogen (*E. coli*) and metals (selection of 11) content are regulated by the Nutrient Management Act as Non-Agricultural Source Material (NASM) (O. Reg. 267/03 – Schedule 5 and 6, CM2 and CP2). Land application is a common practice in Ontario (Jin and Parker, 2017) given that the nutrients within the biosolids (N/P/K) reduce the chemical fertilizer requirements of the crop to which they are applied. The Ontario regulatory environment is different than the US, where EPA guidelines distinguish between a greater variety of pathogen levels (US EPA, 1993). Under EPA guidelines, a "Class A" product must contain less than 1000 MPN/g of *E. coli*, while a "Class B" product must have less than 2×10^6 CFU/g (US EPA, 1993). The latter value is identical to the NASM requirement in Ontario (O. Reg. 267/03 – Schedule 5, CM2). For each plant studied, the nutrient, metal, and pathogen (*E. coli*) content of the biosolids product was evaluated.

Figure 8 shows the mean total phosphorus (TP), total nitrogen (TN), and potassium (K) content of the biosolids products. TP content ranged from 19 – 40 grams of TP per kg of dry solids. The 25th, 50th, and 75th percentile corresponded to 26, 31, and 36 g TP/dry kg, respectively. TN values ranged from 25 – 69 grams of TN per dry kg of dry solids. The 25th, 50th, and 75th percentile corresponded to 39, 44, and 50 g TN/dry kg, respectively. Among the nine facilities that did not add K to their sludge during the treatment process, the product contents ranged from 0.9 – 6.0 grams of K per kg of dry solids. The 25th, 50th, and 75th percentile corresponded to 2.2, 3.3, and 4.0 g K/dry kg, respectively. The facility that did add supplemental K to its sludge (plant H) obtained a biosolids product with 53 g K/dry kg, albeit with a substantial degree of

variability (standard deviation = 26 g K/dry kg). Across all facilities studied, the TP, TN, and K (non-supplemented) contents were broadly consistent with those found in literature (Metcalf & Eddy, 2013).
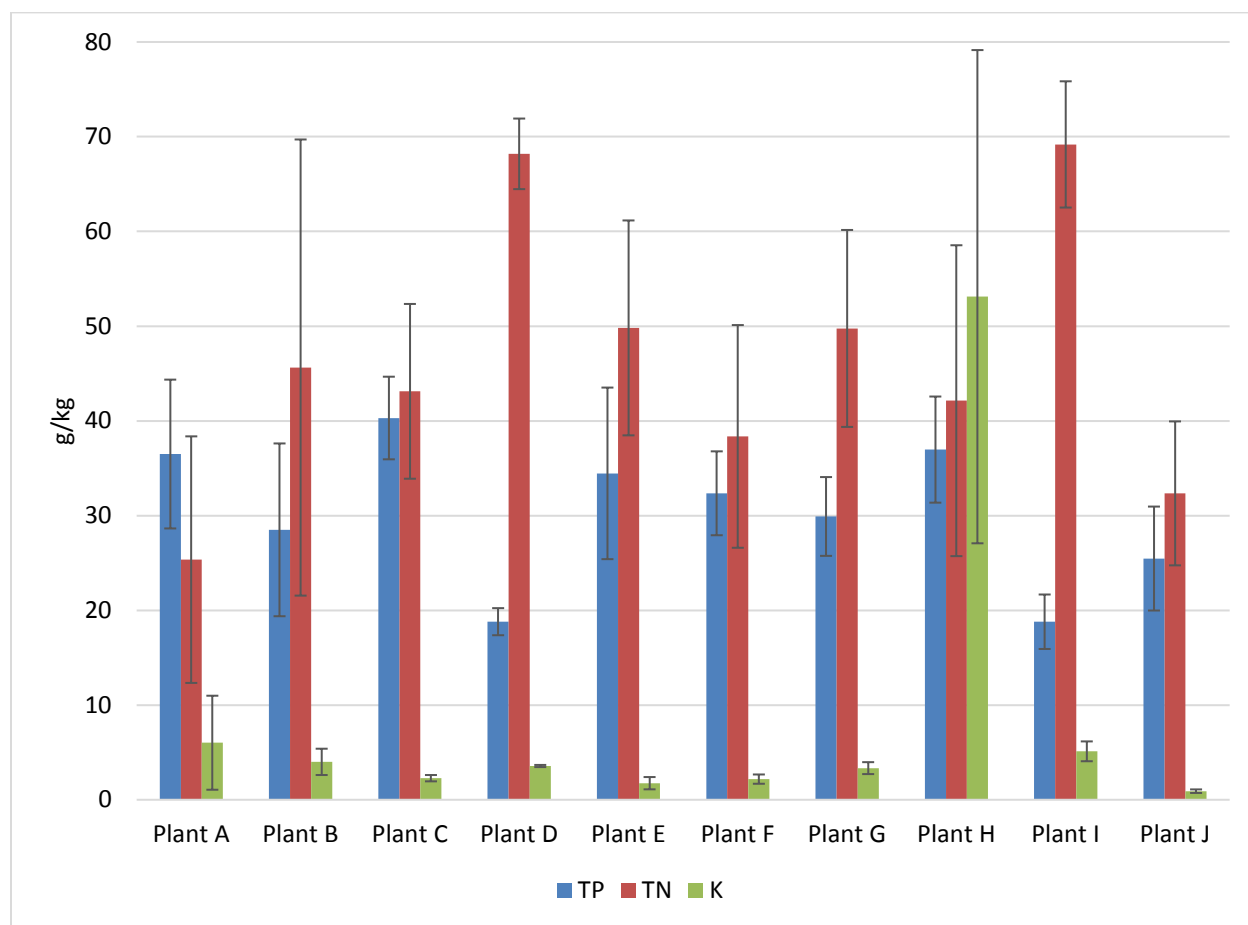
Figure 8: Mean nutrient content of hauled biosolids (dry mass basis)

Figure 9 shows the mean log of the biosolids *E. coli* content for each product evaluated. In all cases, the *E. coli* values were at least one log (*i.e.* 10-fold) lower than the NASM limit of 6.3 log (CFU/g). The content ranged from 2.1 – 5.2 log (CFU/g), while the 25th, 50th, and 75th percentile corresponded to 2.3, 3.8, and 4.4 log (CFU/g), respectively. Four facilities generated a product that met Class A requirements (plants D, E, G, J). This result was expected for plants G and J, since both facilities employed stabilization processes (thermo-alkali hydrolysis and ATAD, respectively) that pasteurized the sludge (Metcalf & Eddy, 2013). The result was more notable for plants D and E, which both employed conventional aerobic digestion. The former facility did not employ any on-site storage, while the latter employed the GeoTube™ process following the digestion process. As a form of long term storage, the GeoTube™ functions similar to a system implemented by Eyre *et al.* (2018), which also generated a Class A product. Taken together, the observations indicate that long-term storage may emerge as a solution for obtaining a Class A product without substantial energy and labour inputs. Nonetheless, given that observed *E. coli* values were at least one log below the NASM limit in each of the case studies, it is clear that pathogen content is not a concern under the current Ontario regulatory framework.
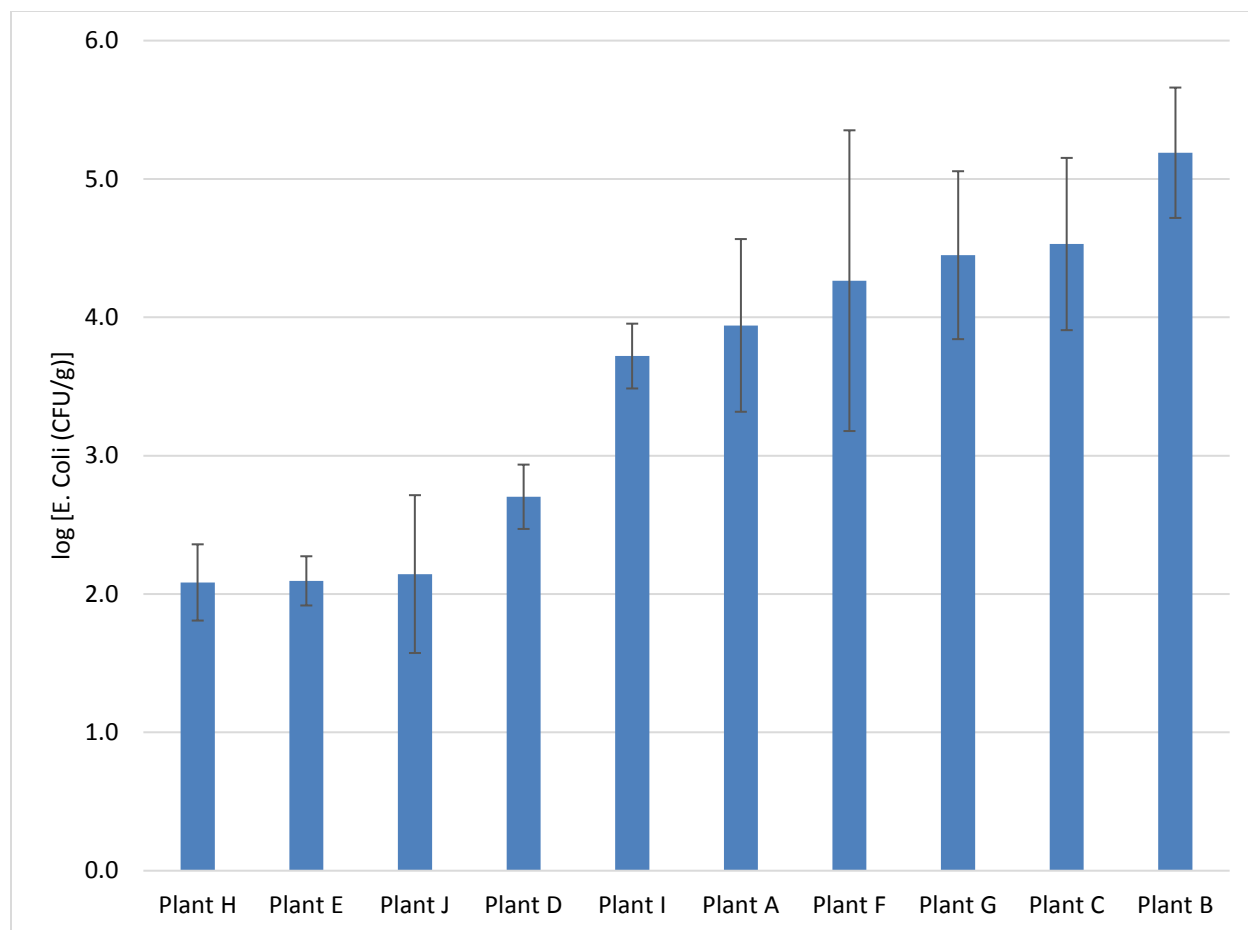
Figure 9: Mean log *E. coli* content of hauled biosolids

The ratio of each metals mean dry weight concentration to the NASM limit was evaluated as a measure of the extent of metal contamination. Figure 10 displays the three highest ratios observed for the plants in the study. From Figure 10, it can be seen that all heavy metal concentrations were below regulatory limits for land application as a NASM. The highest ratio was observed for copper at all plants, and ranged from 0.14 – 0.69 of the NASM limit. Of the copper ratios observed, both the minimum (plant D) and maximum (plant I) occurrences corresponded to the northern facilities that did not practice land application of biosolids. Among the facilities that did practice land application of biosolids, copper ratios ranged from 0.17 – 0.37 of the NASM limit.

Zinc and selenium were the most common metals to be observed as either the second or third highest ratio for the plants evaluated. The former metal was observed in ten occurrences, while the latter was observed in seven instances. The only other metals observed as either the second or third highest ratio were molybdenum (two occurrences) and arsenic (one occurrence). The arsenic observation corresponded to a facility that did not practice land application (plant I).

From the metal content results, it is concluded that metal concentrations are not likely to be an area of concern should more stringent regulations be imposed in the future. Copper exhibited the highest proximity to the NASM limit in each case, but among all plants currently practicing land

application, the copper concentration limit could be reduced by 50% and the biosolids would still be compliant.
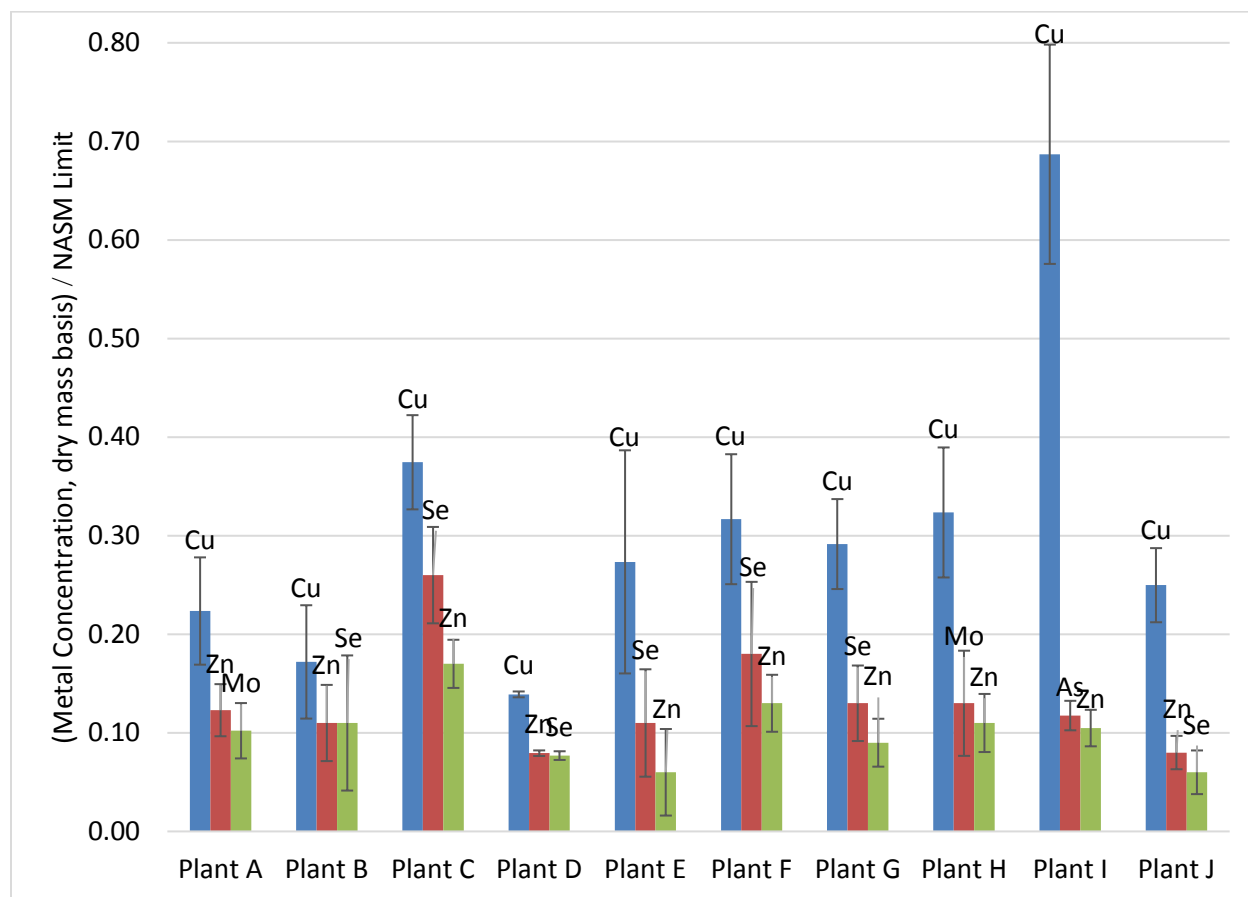
Figure 10: Metals content of hauled biosolids

In summary, all plants in the study exhibited nutrient contents within expected ranges. Median values for TN, TP, and K (non-supplemented) were 31, 44, and 3.3 g/kg, respectively. All facilities met the pathogen (*E. coli*) and metal concentration requirements for application as a NASM. With respect to pathogens, all facilities exhibited at least one log (*i.e.* 10-fold) fewer pathogens than the NASM limit. Four facilities exhibited *E. coli* levels sufficiently low to be classified as a Class A product, one of which was achieved through a low-tech long-term storage technology. With respect to metals, the copper concentration of each product exhibited the closest proximity to the NASM limit. Copper concentrations ranged between 0.17 – 0.37 of the NASM metal limits for plants that currently practice land application. The most common metals corresponding to the second or third highest ratios were zinc (ten occurrences), and selenium (seven occurrences). Overall, the findings for pathogen and metals content indicate that neither are likely to be area of concern should regulations become more stringent.

GHG Emissions KPI Results

To facilitate a comparison of all the sludge-handling operations on the basis of a common metric of environmental sustainability, the inputs and outputs of each system were converted to normalized CO_2 equivalents. The exercise determined the cumulative impact of each input and

output on the carbon footprint of the plant. Where possible, emission factors specific to Ontario (electricity, natural gas production) or Canada (transportation fuel) were used to calculate the emissions associated with each input. In other cases, literature values for chemical production (polymer, KOH) and chemical fertilizer production (N, P, K) were utilized. The latter factors were used to determine the carbon off-sets gained by using biosolids as a fertilizer through the avoidance of chemical fertilizer production for each nutrient.

The contributions of each system input and output to the net carbon footprint of each plant are shown in Figure 11. Net GHG emissions ranged from -119 to 299 kg CO_2 equivalents per dry tonne of raw sludge produced. Of the eight plants that practiced land application disposition, six exhibited net negative emissions, which ranged between -119 and -4 kg CO_2 eq./dt. In each case, the carbon credits gained from chemical fertilizer offsets exceeded the emissions associated with plant operations and biosolids trucking. The outcome was due in large part to the reduction in carbon emissions associated with electricity production in Ontario, which have dropped over 75% since 2010 (Environment and Climate Change Canada, 2018). From a sustainability perspective, the outcome was noteworthy in two respects: a) land application is the most common disposal method for small WWTPs in Ontario (Jin and Parker, 2017), and b) all observations corresponded to facilities practicing conventional aerobic digestion, which is the most common stabilization technology among small WWTPs in Ontario (Jin and Parker, 2017). As such, if comparable facilities were to exhibit inputs and outputs similar to those within the study, it could be concluded that the sludge-handling systems of a number of plants province-wide may be entirely sustainable with respect to GHG emissions. The finding is noteworthy for those seeking insight into best residuals management practices, however, additional analysis of liquid stream operations would be necessary to evaluate the net GHG impact of the entire treatment plants.

Of the plants studied, two facilities practiced land application but did not exhibit negative emissions. A third facility (plant H) exhibited negative emissions (-4 kg CO_2 eq./dt) to a substantially lesser degree than other such plants within the sample [-119 – (-) 86 kg CO_2 eq./dt]. Plant B exhibited emissions near zero (4 kg CO_2 eq./dt), while the emissions associated with plant J (128 kg CO_2 eq./dt) were substantially higher than its peers that practiced land application. The net positive emissions observed for plant B were primarily a result of the trucking emissions (137 kg CO_2 eq./dt), which exceeded those associated with electricity consumption (109 kg CO_2 eq./dt). In the case of plant J, emissions associated with energy consumption and chemical use exceeded the credits gained from fertilizer off-sets.

Among the plants that employed polymers for dewatering, the carbon intensity impact was pronounced for facilities that were identified as using chemical quantities that were greater than those reported in the literature (plants J, D, H). Notably, carbon emissions associated with polymer use was similar to electricity-associated emissions for plants D and J. Conversely, the plants that used polymer quantities consistent with literature values had the second (plant E) and third (plant C) lowest carbon footprint of all facilities studied. The result for plant C was particularly notable in that the facility exhibited the highest distance required for trucking among all plants studied.
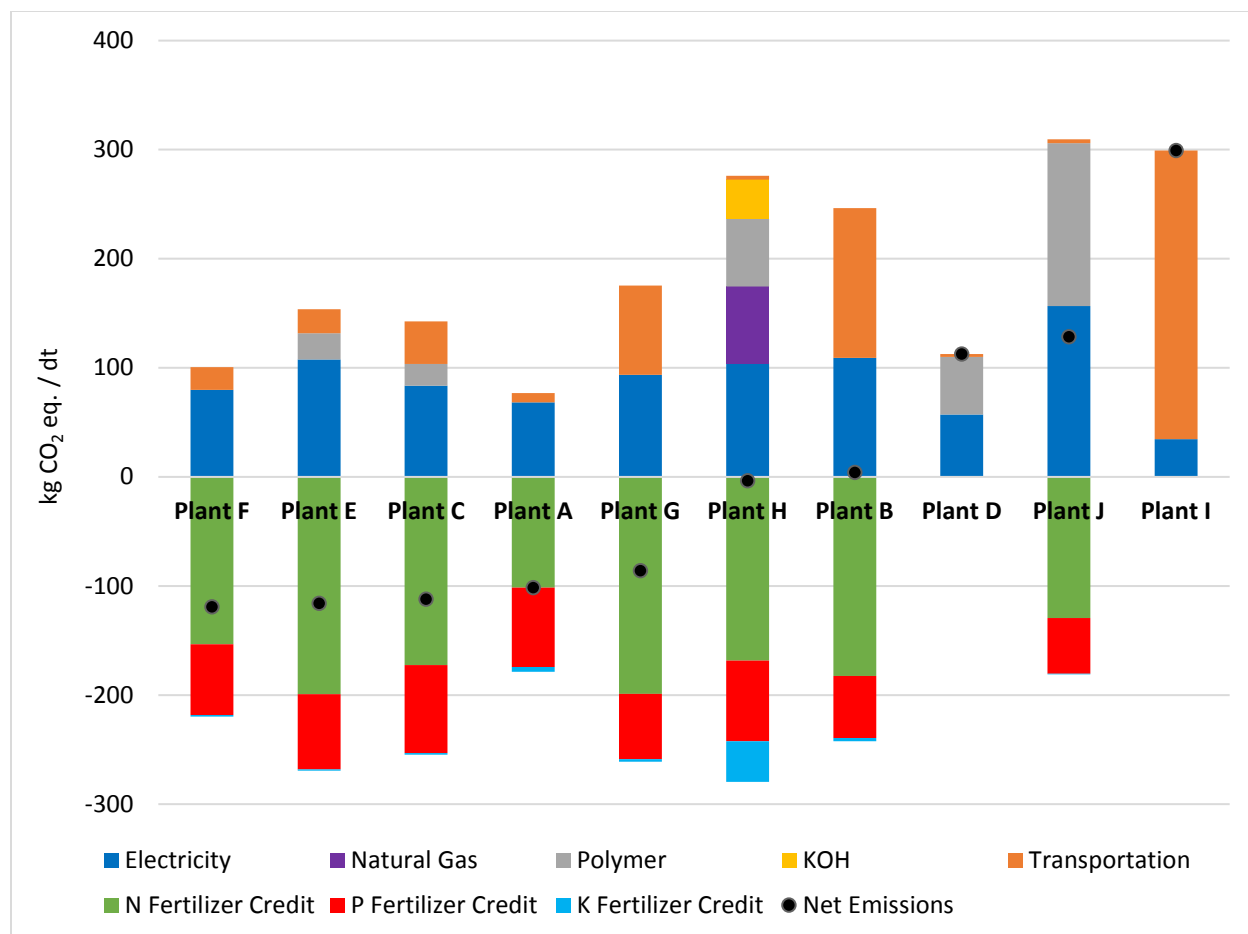
Figure 11: Sludge-handling process GHG emissions per dry mass of raw sludge produced

The highest net quantity of carbon emissions was associated with plant I (299 kg CO_2 eq./dt), which did not receive any carbon credits (biosolids were landfilled). It exhibited 186 kg CO_2 eq./dt more emissions than the other plant (D) that landfilled its biosolids. The latter plant practiced on-site dewatering, thereby providing an indication of the environmental benefits that such a technology can provide. Plant H received a larger K fertilizer credit (-37 kg CO_2 eq./dt) than its peers, which off-set the emissions associated with the production of the KOH chemical used in the stabilization process (36 kg CO_2 eq./dt). Its electricity-related emissions totalled 104 kg CO_2 eq./dt, of which 90 kg CO_2 eq./dt were collectively associated with aerated holding and odour control. Therefore, if the technology were to be implemented at facilities with no aerated holding or odour control requirements (but with similar trucking distance), the technology would exhibit emissions similar to the other observed sludge-handling systems that were carbon sinks.

Figure 12 shows the carbon emissions associated with processes upstream of and including stabilization for each plant in the study. The analysis provides insight into the emissions required for mechanical treatment processes to achieve the observed product quality (pathogen content), irrespective of downstream processes. As noted previously, plants D, J, and H generated a Class A product directly as a result of their mechanical stabilization process (plant E generated it via passive long-term storage). However, only the latter two facilities generated this quality of product using thermal processes that have historically been accepted as being able to consistently

generate a Class A product (Metcalf & Eddy, 2013). Plant D employed conventional aerobic digestion, which would be expected to produce a Class B product (Metcalf & Eddy, 2013). From Figure 12, it can be seen that the highest emissions were associated with plants H and J, which highlighted the environmental cost (GHG impact) of obtaining a Class A product through thermal mechanical processes.
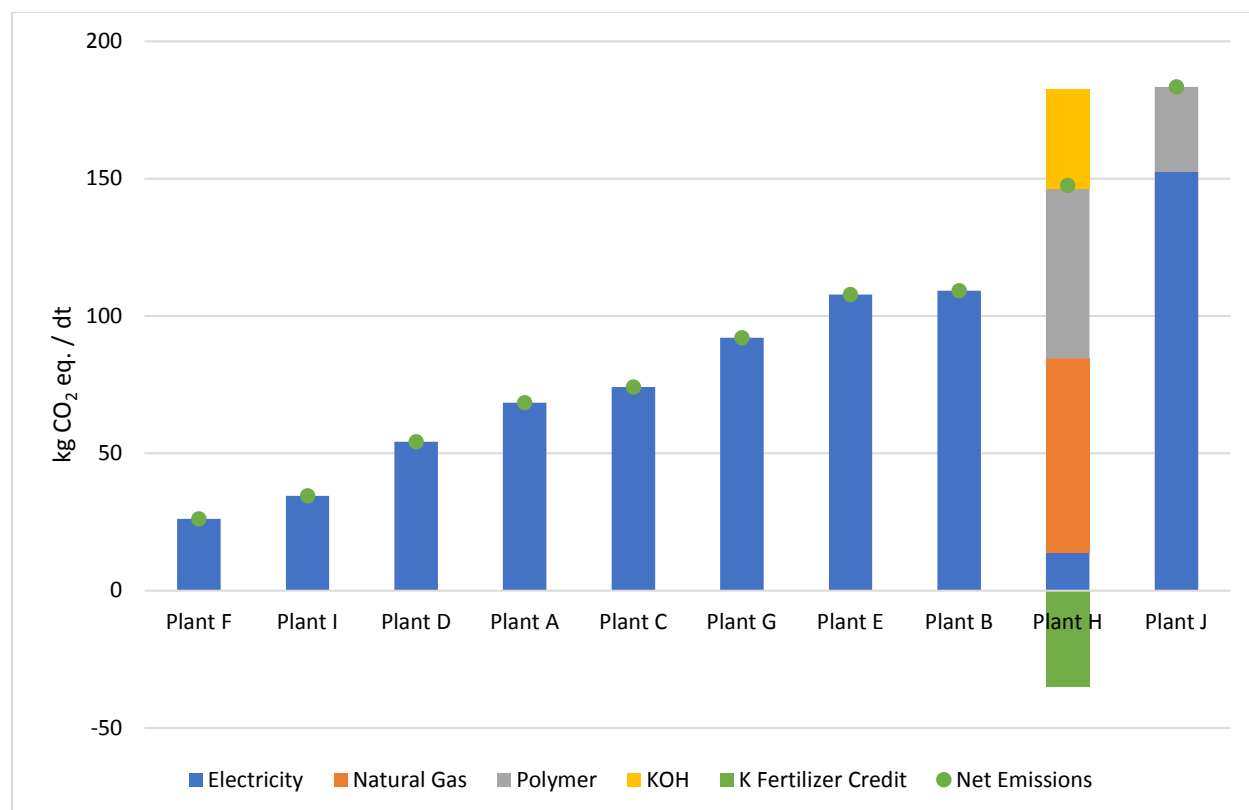
Figure 12: GHG emissions associated with processes upstream of and including stabilization

In the case of plant J, both the thickening and stabilization steps were necessary to generate the Class A product. Emissions associated with pumping, thickening, and stabilization electricity consumption (152 kg CO_2 eq./dt) and thickening polymer usage (31 kg CO_2 eq. / dt) summed to 183 kg CO_2 eq./dt. For plant H, electricity associated with pumping, stabilization, and upstream dewatering (14 kg CO_2 eq./dt), natural gas (71 kg CO_2 eq./dt), polymer usage (62 kg CO_2 eq./dt), and KOH (36 kg CO_2 eq./dt) were necessary to generate the final product. The addition of KOH generated a carbon credit of -35 kg CO_2 eq./dt (credits for N and P fertilizer avoidance were not a result of the treatment process selection). Subtracting the credits associated with the K fertilizer avoidance, the emissions associated with the inputs identified totalled 148 kg CO_2 eq./dt. Examining exclusively stabilization and related emissions, both plant J and H exhibited emissions substantially higher than the plants that generated a Class B/NASM quality product. The emissions associated with Class B generation ranged between 26 – 109 kg CO_2 eq./dt.

In summary, GHG emissions associated with sludge-handling operations ranged from -119 to 299 kg CO_2 eq./dt among all plants studied. Six facilities exhibited net-negative emissions, which ranged between -119 to -4 kg CO_2 eq./dt. The five systems that yielded the lowest

emissions employed process configurations that were relatively common province-wide, which suggests that other sludge-handling systems across the province may be carbon sinks as well. Among the two plants (H and J) that practiced thermal stabilization processes (thermo-alkali hydrolysis and ATAD) for the purposes of generating a Class A product, total emissions associated with stabilization and auxiliary processes were 148 and 183 kg CO_2 eq./dt, respectively. Both values were higher than plants that generated Class B/NASM quality products, which ranged between 26 and 109 kg CO_2 eq./dt. The discrepancy highlighted the environmental trade-offs associated with Class A product generation through such stabilization methods.

CONCLUSIONS

A benchmarking exercise was completed to characterize the sustainability of sludge handling in small WWTPS in Ontario. Ten plants across the province were evaluated on a variety of sustainability metrics: energy consumption, chemical use, biosolids disposition, biosolids quality, and GHG emissions. The metrics were chosen to capture all the inputs and outputs of the sludge-handling process: treatment, transportation, and end-use.

Among all plants studied, overall electricity consumption for sludge-handling ranged from 0.86 – 3.9 kWh/dry kg (median = 2.2 kWh/dry kg). The maximum consumption corresponded to the facility practicing ATAD stabilization, while the highest value among conventional aerobic digestion plants was 2.7 kWh/dry kg. Consumption for stabilization processes was found to range from 0.25 – 3.8 kWh/dry kg (median = 1.6 kWh/dry kg). The maximum value corresponded to the ATAD process, while the minimum consumption corresponded to the thermo-alkali hydrolysis process. The low value for the latter process indicated that such a technology may be a viable option for reducing electricity consumption in facilities where aerated holding and odour control are not necessary. The high consumption associated with the ATAD plant suggested that such a process is energy-intensive relative to conventional processes.

Among the eight plants that practiced conventional aerobic digestion, consumption normalized by quantity of VSS destruction was found to range from 4.9 – 56 kWh/dry kg VSS (median = 8.7 kWh/dry kg VSS). The ATAD facility exhibited the highest consumption among all plants (63 kWh/dry kg VSS). Of the eight plants that practiced conventional aerobic digestion, the range of the five lowest values (5 – 9 kWh/dry kg VSS) was substantially lower than the range of the three highest values (15 – 56 kWh/dry kg VSS).

Electricity consumption for mechanical dewatering processes ranged from 0.06 – 0.10 kWh/dry kg, which represented 2 – 5% of total sludge-handling power draw for such plants. Chemical usage for dewatering processes ranged between 8 – 24 kg polymer/dt (centrifuges) and 20 – 28 kg polymer/dt (rotary presses), while the GeoTube™ process used 9 kg polymer/dt. The three highest observed chemical usage values were greater than those found in literature, which suggested that the polymers were either over-dosed to some extent and/or were not as effective coagulating/flocculating agents as those employed in literature case studies. The solids content of the product generated ranged between 16.9 – 22.5% (centrifuges) and 17.2 – 18.4% (rotary presses). The GeoTube™ process generated a 9.2% solids product.

The weighted average round-trip distance for disposition between the WWTP and the final destination ranged between 8 – 83 km and the normalized transportation fuel consumption ranged between 1 – 99 L/dt. The difference between trucking distance and transportation fuel consumption was largest when comparing facilities that employed chemically enhanced dewatering against those that did not. Among three facilities that exhibited the highest disposition distance, only the plant with the maximum trucking distance (83 km) practiced on-site dewatering. Despite the high trucking distance exhibited, it consumed between 72 – 85% less fuel than the two other plants of interest.

All the sampled plants exhibited nutrient contents within expected ranges and contaminant (metals/pathogens) contents below regulated levels for application as a NASM. All facilities exhibited at least one log (*i.e.* 10-fold) fewer pathogens than the NASM limit. Four facilities generated Class A product with respect to pathogen content, one of which was achieved through a low-tech long-term storage technology (GeoTube™). The copper concentration of each product exhibited the closest proximity to the NASM limit. It ranged between 0.17 – 0.37 among plants that currently practiced land application. The most common metals corresponding to the second or third highest ratios were zinc and selenium. Overall, the findings for pathogen and metals content indicate that neither are an area of concern should regulations become more stringent.

Carbon emissions ranged from -119 to 299 kg CO_2 eq./dt among all plants studied. Six facilities exhibited net-negative emissions, ranging from -119 to -4 kg CO_2 eq./dt. Five of the six systems that yielded negative emissions employed process configurations that were relatively common province-wide, which suggests that a number of other sludge-handling systems across the province may be carbon sinks as well. Among the two plants that practiced an alternative stabilization processes (thermo-alkali hydrolysis and ATAD) for the purpose of generating a Class A product, emissions associated with stabilization and auxiliary processes were 148 and 183 kg CO_2 eq./dt, respectively. Both quantities were substantially higher than those associated with processes that generated Class B/NASM quality products (26 – 109 kg CO_2 eq./dt) and highlighted the environmental trade-offs associated with generating Class A products through such stabilization methods.

From an operations and sustainability standpoint, the benchmarking approach developed can be applied by plant owners and operators who seek to better understand how their utility is performing relative to peers of similar capacity and scope of operations, identify areas of need and further investigation, and ultimately improve the long-term sustainability of their operations. The substantially different levels of normalized inputs/outputs observed demonstrate the value of benchmarking: it provides owners and operators with a means to compare, evaluate, and potentially find opportunities for optimization within their own systems.

ACKNOWLEDGEMENTS

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