Research and Technical Analysis to Support and Improve the Alternative Manure Management Program Quantification Methodology

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Disclaimer

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Abstract

Default factors related to manure deposition and solids separation currently used in the Alternative Manure Management Program (AMMP) quantification methodology (QM) for California livestock operations were evaluated and recommendations for improvement are provided. Topics reviewed and analyzed include: time budgets for cows and manure deposition locations, solid separators, weeping wall systems, aerobic treatment (lagoons), and gasification systems.

Executive Summary

This analysis evlautes and recommends improvements to default emission factors currently used the California Air Resources Board's (CARB) Greenhouse Gas (GHG) Quantification Methodolgy for the California Department of Food and Agriculture (CDFA) Alternative Mamure Management Program (AMMP)[1] The report consists of five chapters, one for each task in the scope of work which include:

1. Technical Review of Default Factors for Manure Deposition in Quantification Methodology for Dairy and Livestock Operations
2. Technical Review of Solids Separation Default Factors and Recommendations for Addition of New Factors
3. Assessment of Methane Emissions from Weeping Wall Solids Separation Systems
4. Assessment of Commercially Available Aerobic Treatment Systems, and Recommendations Regarding Research Needs to Evaluate an Aerobic Treatment System
5. Evaluation of Livestock Manure Pyrolysis and/or Gasification Systems

Chapter summaries with recommendations appear below.

Chapter 1: Technical Review of Default Factors for Manure Deposition in QM for Dairy and Livestock Operations

The AMMP Quantification Methodology uses default factors for the fraction of manure deposited on land and/or in corrals for specific dairy animal class (i.e., lactating cows, dry cows, heifers, and calves) and housing type (i.e., freestall, open lot, pasture). This "deposited on land" fraction represents manure that is not collected as liquid or slurry and then sent to anaerobic storage. They are based on estimates for the average amount of time livestock spends in a given area of the farm (i.e., freestalls and feed lanes, open-lot corral, milking parlor, etc.). The "deposited on land" default values in the Quantification Methodology for lactating cows are 20%, 70% and 90% for freestall, open lot, and pasture housing/management, respectively. For dry cows, the Quantification Methodology default value is 70% deposited on land (i.e., 30% recoverable).

Time budgets in the literature for lactating dairy cows in the U.S. Midwest and South-Central Turkey, suggest recoverable manure fractions of 80-100% for animals in freestall housing (no open lot access) and about 50% for those with access to open lots. Results from a recent study at four California dairies suggest 82-90% (87% range midpoint) and 43-49% (46% midpoint) recoverable manure fraction for lactating cows in freestall and open lot access housing, respectively. For dry cows, the California study suggests the manure recovery fraction is 21-36% (29% midpoint).

The recoverable manure fraction for lactating cows from the recent California literature is slightly higher for freestall housing than the AMMP Quantification Methodology default (87% and 80%, respectively) and likely significantly higher for open lot housing (46% and 30%, respectively).

[1] CDFA AMMP website: https://www.cdfa.ca.gov/oefi/AMMP/

For dry cows, the California based data appears to confirm the AMMP Quantification Methodology default of 30%.

Though recent time-on-concrete data suggest higher default recovery rates for lactating cows, with concomitant higher GHG emissions, there are not enough data (or dairies evaluated) to justify a change to the default recovery rate at this time.

Recommendations:

No change to AMMP Quantification Methodology manure recovery defaults is recommended at this time, though more data are needed.

For dry cows, the measured time-on-concrete yields manure recovery close to the Quantification Methodology default. A moderate deviation from the Quantification Methodology default for this class of animal would have relatively small impact on the overall manure recovery value calculated by the Quantification Methodology because dry cows comprise only about 15% of the adult herd and excrete less than lactating cows due to lower feed intake.

While the time-on-concrete data for lactating cows suggests higher default recovery rates, there are not enough data (or dairies evaluated) to justify a change.

Because manure recovery rates play such an important part in the GHG estimates in the Quantification Methodology, more time-on-concrete data collection is recommended from a variety of dairies (i.e., large, small, freestall with and without access to open lot).

Technology for monitoring health, productivity, and on-farm location of individual cows has advanced in the past 10 years. Activity sensors embedded in ear tags or ankle monitors, with appropriate router set-up, allow 24/7 monitoring (including location) of each cow on the dairy.[2] Further research should be explored that utilizes location-activity monitors to record and analyze data from a wide range of dairies in the State. Careful experimental design would be needed with sufficient justification that the dairies analyzed are representative.

Chapter 2: Technical Review of Solids Separation Default Factors and Recommendations for Addition of New Factors

Most California dairies use some method of solids separation. Approximately, 30-40% use settling ponds or basins only and approximately 30% use mechanical separation, with or without settling basins. Sloped screen separators are by far the most common mechanical separators used on California dairies, followed by drag flight conveyor separators (also known as scraped screen separators). It is estimated there are fewer than ten screw press and weeping wall system separators and no roller press or centrifuge separators employed on California dairies (Meyer, 2019).

This analysis reviews and analyzes dairy manure solids separation efficiencies for the following mechanical separator types: sloped screen, two-stage sloped screen, drag flight conveyor, rotary drum, centrifuge, screw press, and roller press. Weeping wall systems are addressed in Chapter 3.

Bedding material commonly used on California dairies includes recycled manure solids, nut shells, rice hulls, straw or other. Some of this bedding material, as well as spilled food rations, are flushed with manure adding to the solids and volatile solids (VS) load sent to liquid storage.

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[2] E.g., see Smartbow (https://www.smartbow.com/en/home.aspx), Cowmanager (https://www.cowmanager.com/en-us/Solution/Modules#findmycow), and others.

Some 2-10 kg of bedding VS is added per cow per day according to Arndt et al. (2018), and Meyer, et al. (2019). In the steady state, with no overall accumulation of bedding material in the freestall housing, an amount equivalent to that added is likely pushed into the feed lanes and flushed along with manure. Neither the AMMP Quantification Methodology nor the CARB's Compliance Offset Protocol for Livestock Projects (CARB, 2014) account for the impact of bedding material on GHG emissions estimates. Some research has reported an influence of bedding VS on GHG emissions from stored manure while other work recognizes the potential and recommends improved VS accounting and evaluation of bedding influence on emissions.

As more bedding and/or feed material is flushed with manure, the total VS load sent to a separator increases. Separators, more or less, remove a proportion of the incoming solids from the liquid manure. A higher solids input will yield more separated solids as well as more solids in the liquid effluent. A simple model was developed in this analysis to estimate the effect of bedding on effective separator efficiency.

Recommendations:

Table ES-1 summarizes recommendations for "nominal" default separator efficiencies in the AMMP Quantification Methodology. In concurrence with Meyer, et al., (2019), it is also recommended that CARB evaluate the effect of other VS sources and flows, that are flushed with raw manure (e.g., bedding, feed, recycled manure solids), on effective separation efficiency and lagoon methane emissions. If deemed significant, these other VS flows should be accounted for in the Quantification Methodology.

Table ES-1. Summary of Recommendations for Separators

Separator Type	**Recommendation**	**Current AMMP Quantification Method Default (Solids Removal)**
Sloped screen	**30-35% default solids removal**	17%
Two-stage sloped screen	**50% default solids removal** (based on two studies with consistent results)	**None**
Drag flight conveyor or scraped screen	Conduct measurements on 2-3 drag flight conveyor separators in California and derive a QM default for this class of separator.	**None**
Rotary Drum Separator (Roller Drum)	No change	25%
Centrifuge	No change	50%
Screw press	**25% Default for Flush Systems 50% Default for Scrape/Vacuum Systems**	25%
Roller Press (Belt Press)	No change	50%
Weeping Wall	Addressed in Chapter 3	45%

Chapter 3: Assessment of Methane Emissions from Weeping Wall Solids Separation Systems

A weeping wall system is an engineered basin with one or more perforated walls that form a large dewatering surface area. Compared with mechanical separation technologies, a weeping wall offers several advantages including lower energy consumption, repair, and maintenance cost. They provide flexibility in manure management with extended storage periods for manure solids (3 to 9 months).

Generally, weeping wall systems consist of at least 2 rectangular cells, each as a standalone structure with a concrete floor and one to three sides made of perforated walls (i.e., slotted concrete or screens supported by steel or concrete pillars) and are typically 5 to 8 feet tall. The fourth side is generally solid and sloped and used as an entry ramp for filling and for equipment used to excavate (empty solids from) the cell.

Low solids flushed manure is directed to the basin entry and travels along the cell. Liquid drains through the perforated "weeping" walls as solids accumulate inside the cell. The drained liquid usually flows to and is stored in lagoons for use as flush water and/or irrigation. The accumulated solids in the cell act as a filter that helps capture more solids. Depending on cell volume and number of animals, the cell can take 2 to 16 weeks to fill after which inlet flow is directed to an empty cell. The full cell is left to continue to drain for weeks to months while functioning as manure storage. After storage, the solids are removed and then composted or spread on nearby fields.

Weeping wall systems undoubtedly experience anaerobic conditions for much, if not all, of their cycle (fill, drain/dry, and excavate). Significant methane emissions from a weeping wall system and anaerobic stacked manure (an analog to weeping wall storage) have been measured in relatively cool climates (i.e., New Zealand and the United Kingdom). Emissions would likely be higher in the warmer climate of the San Joaquin Valley.

Though weeping wall systems remove 50-80% of solids from liquid manure flow to a storage lagoon, reducing methane emissions from the lagoon, they also likely emit methane during the months long "separation and solids storage" interval before excavation to composting or land application. Without relevant emissions measurement and modeling from weeping wall systems in California, this analysis developed a simple model based on analogous system emission factors including stage-specific methane conversion factors and relative duration. The model estimates an effective weeping wall system methane conversion factor of 0.22 for retained solids.

Recommendations:

A 65% solids retention default with a methane conversion factor (MCF) of 0.22 is recommended for weeping wall systems in the Quantification Methodology. The current default is 45% solids retention with no methane emissions (0 MCF)). If adding the recommended MCF, or similar, to the Quantification Methodology is not acceptable, then no change to the weeping wall solids retention default is recommended.

In addition, a weeping wall measurement and modelling research program is recommended to obtain a more complete understanding of the system emissions.

Chapter 4: Assessment of Commercially Available Aerobic Treatment Systems, and Recommendations Regarding Research Needs to Evaluate an Aerobic Treatment System

Aeration techniques common at municipal wastewater treatment facilities have been used in swine and dairy manure management, primarily for odor control, for many years. Lab based studies demonstrate that the five-day biochemical oxygen demand (BOD_5) or VS are reduced by aeration of dairy and pig manure (by > 96% in one study), which implies lower potential methane emissions. Aeration techniques at swine facilities were measured (or modeled) to reduce overall GHGs by approximately 55% in simple open aerated tanks, to more than 99% for a sophisticated aerobic treatment system. Though more nitrous oxide (N_2O) is emitted by aerated treatment devices than from anaerobic lagoons, overall system N_2O emissions are likely lower because there is less nitrogen in the treated material used in land application (i.e., lower N_2O emissions from land application at least for treated swine manure). Complete aeration (oxidation) of an anaerobic lagoon is energy intensive (estimated electricity cost of approximately $550 per cow per year) primarily because of the complete-mix requirement and large volume of a storage lagoon. It is unclear whether surface aerators, used for odor control, can provide sufficient mixing.

A research study to measure emissions from an aerated lagoon is achievable using ground-based area source measurement techniques. A preliminary cost estimate for this research is about $450,000, not including the cost of an aerator system and any needed lagoon modification.

Recommendations:

There is not enough information regarding the performance of aeration systems in dairy lagoons to warrant updates to the AMMP Quantification Methodology at this time. A research study to measure and model emissions from aerated lagoons is recommended. This should include monitoring nitrogen fate and benefits or impacts due to changes in nitrogen in the land-applied treated manure.

Chapter 5: Evaluation of Livestock Manure Pyrolysis and/or Gasification Systems

Conversion of organic material can proceed along three main pathways—biochemical, thermochemical, and physicochemical.

Biochemical conversion processes include anaerobic digestion (AD) and/or fermentation, which produces methane and carbon dioxide (CO_2), and aerobic conversion such as composting, which produces a more or less stabilized organic material and CO_2. Biochemical conversion proceeds at lower temperatures and lower reaction rates which can require large volume reactors. Higher moisture feedstocks are generally good candidates for biochemical processes.

Thermochemical conversion processes include combustion, gasification and pyrolysis. Thermochemical conversion is characterized by higher temperature and conversion rates and is best suited for lower moisture feedstocks. Combustion is the complete oxidation of the fuel at elevated temperatures for the production of heat without generating commercially useful intermediate fuel gases, liquids, or solids. Gasification refers to the conversion of a solid or liquid feedstock into an energetic, or fuel, gas (often called producer gas or synthesis gas). Autothermal gasification uses partial oxidation of the substrate to produce principally carbon monoxide (CO), hydrogen (H_2), methane (CH_4), and light hydrocarbon gases in association with CO_2, molecular nitrogen (N_2) and water vapor. Allothermal (indirectly heated) gasification uses an external heat source for the gasification reactor. Products also include liquids (tars, oils, and other condensates) and solids (char, ash). The fuel gases can be used in internal and external combustion engines, fuel cells, and other prime movers, or as a chemical feedstock for other products including liquid fuels. Pyrolysis is similar to gasification except no added air or oxygen is used and it is generally optimized for the production of fuel liquids (pyrolysis oils) or char solids (biochar).

Evaporation or solid/liquid separation with follow-on solids drying is required to obtain a suitable feedstock for most thermal conversion systems. This separation or evaporation activity would be responsible for any GHG reduction due to diversion from lagoon. The gasifier (or thermal conversion) would be responsible for any additional GHG reduction due to its products displacing fossil or non-renewable products, such as natural gas combustion or grid electricity.

Recommendations

We do not recommend including gasification or thermal conversion in the AMMP Quantification Methodology.

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1. Technical Review of Default Factors for Manure Deposition in the Quantification Methodology for Dairy and Livestock Operations

Background

Default factors are used in the Alternative Manure Management Program (AMMP) Quantification Methodology for the fraction of manure deposited on land and/or in corrals. This fraction cannot be collected as liquid or slurry and then sent to anaerobic storage. Factors are based on estimates for the average amount of time livestock spends in a given area of the farm (i.e., freestalls and feed lanes, open-lot corral, milking parlor, etc.) (Table 1-1).

Table 1-1. Percent manure deposited on land and percent recoverable default factors for the AMMP Quantification Methodology

	Manure deposited on land (%)	**Recoverable (%)**
Lactating Dairy Cows (freestall)	20%	80%
Lactating Dairy Cows (open lot)	70%	30%
Lactating Dairy Cows (pasture)	90%	10%
Cattle: dry cows	70%	30%
Cattle: heifers (on feed)	70%	30%
Cattle: bulls	100%	-
Cattle: calves (grazing)	100%	-
Cattle: cows (grazing)	100%	-
Cattle: heifers (grazing)	100%	-

The Quantification Methodology uses an average daily time budget for dairy cows from a report by a University of California Committee of Experts on Dairy Manure Management (Chang, Harter, & Meyer, 2005). Regarding “partitioning of manure on dairy surfaces", Chang et al. (2005) states “no published studies exist to describe manure production in each area on a dairy.” The report assumes that excretion is uniform throughout the day and that deposition by area is therefore proportional to time spent by area (Table 1-2).

Table 1-2. Average daily time budget for California dairy cows and recoverable manure fraction (hours per day)

Housing Type	**Corral/ Pasture**	**Stall/Alley (includes feeding)**	**Milking**	**Total Concrete Surfaces***	**Recoverable Fraction (via flush)**
Pasture	19.5-22		2-4.5	2-4.5	8-19 %
Freestall w/ corral	12-19.5	3-7	2-4.5	5-11.5	21-48%
Freestall-no corral		8-19.5	2-4.5	10-24	42-100%

*Assumes Stall/Alley and Milking Parlor are on concrete flushed surfaces.
Source: (Chang et al., 2005)

Time Budget Literature Review

Most of the literature addressing dairy cow time budgets accounts for time spent feeding, milking, standing, socializing and lying (resting and ruminating). For example, Grant (2009) discusses time budgets for cows housed in freestall barns without access to loafing pens or corrals (Table 1-3). With 3.5 hours per day spent milking, the remaining 20.5 hours is spent in the barn feeding, standing and resting. Cows spend about 50% of the day lying down (Grant, 2009). Grant and Cook (2010) analyzed time budgets for 205 lactating cows housed in 16 freestall barns in Wisconsin (no access to open lots) using data collected in Cook et al., (2004, 2007, 2008). Time outside the pen for milking (travel, waiting to enter the milking parlor, and milking) averaged 2.7 hours while all other activities (feeding, other time in alley, standing and lying in stall) summed to 21.3 average hours (Table 1-3). To the extent that the dairy has concrete walkways to, and waiting stall before, the milking parlor, recoverable manure for cows in a freestall barn without access to open lots is 80-100%.

Empirical literature under conditions roughly similar to California is limited (see discussion on new study by Meyer et al. (2019) below). Seyfi (2013) and Seyfi & Ugurlu (2010) detail observations of a dairy herd with freestall housing along with access to a coral. The herd was in Konya located in South-Central Turkey. The elevation is 3,300 feet with average temperature of 25-30 °F and 85-90 °F in winter and summer, respectively (Turkish State Meteorological Service (2019)). They observed that cows chose to spend 12.25 hours per day in the coral, overall on an annual average (15 hours/day in spring and summer and 6 hours/day in the winter). The Seyfi papers suggest that in a climate with cold winters and mild summers, approximately 50% of manure is recoverable (via flush) for cows with freestall housing and open access to a corral.

Table 1-3. Dairy cow time budgets from the literature (hours per day)

Housing Type	Corral/ Pasture	Stall/Alley (includes feeding)	Milking	Total Concrete Surfaces	Recoverable Fraction (via flush)	Sources
Freestall		20.5-21.3	2.7-3.5	~24	80-100%	Grant (2009); Gomez & Cook (2010)
Freestall w/ access to open lot	12.25	9.75	2	11.75	~50%	Seyfi (2013); Seyfi & Ugurlu (2010)

Meyer et al., (2019) observed lactating and dry cows at four California dairies in order to determine the average amount of time an animal spends "on concrete" where the manure is regularly collected via flush, scrape or vacuum methods, versus open lot (natural surface), pasture, or other uncovered surfaces where collection is less frequent or not at all (Table1-4). Two of the dairies utilized freestall housing with access to open lot, and two were open lot with shade structures and shaded feed lanes. Population at the four dairies ranged from 2,800 to 5,300 mature cows. Three 24-hour observations were conducted at each dairy (one each in spring, summer, and winter) during which the fraction of cows in a sentinel pen standing on the concrete surface was periodically tabulated. The overall average of these tabulations represents average time on concrete in a 24-hour period. The Meyer et al., (2019) data does not appear to account for time spent milking (travel to and from milking parlor, waiting in paved crowding areas prior to milking, and actual in-parlor time). Therefore, average milking time from the literature (2.8 hours per day) was added to the time on concrete values.

Lactating cows spend 19.6-21.5 and 10.2-11.7 hours per day on concrete at freestall and open lot dairies, respectively (Table 1-4). Based on these time on concrete observations, the recoverable fraction of manure, via flushing, from lactating cows is 82-90% and 43-49% at freestall and open lot dairies, respectively. These are somewhat higher than the default recoverable fractions in the AMMP Quantification Methodology for lactating cows, which are 80% and 30% for freestall and open lot housing, respectively (Table 1-1).

Dry cows spend considerably less time on concrete because they do not visit the milking parlor and spend less time eating. For dry cows, observed time on concrete was similar at both freestall and open lot dairies (between 5 and 8.6 hours per day). Recoverable fraction of manure for dry cows ranges from 21-36% which supports the AMMP Quantification default of 30% recoverable (Table 1-1).

Table 1-4. TOC summary recoverable fraction of manure for lactating and dry cows at four California dairies.

Housing Type	**Open Lot (h/d)**	**Stall/Alley [includes feeding] (h/d)**	**Milking (h/d)**	**Time on Concrete (h/d)[d]**	**Recoverable Fraction (%, via flush)**
Lactating Cows					
Freestall[a]	2.5-4.4	16.8-18.7	2.8[c]	19.6-21.5	82-90%
Open Lot[b]	12.3-13.8	7.4-8.9	2.8[c]	10.2-11.7	43-9%
Dry Cows					
Freestall[a]	15.4-19	5-8.6	-	5-8.6	21-36%
Open Lot[b]	18-18.3	5.7-6	-	5.7-6	24-25%

Notes: a. With access to open lot.
b. With flushed (concrete) feed lanes.
c. Average milking time of 2.8 hours was added to TOC values from Meyer et al. (2019). The 2.8 hours is an average of times from Beggs et al., (2018), Charlton et al., (2017), Charlton et al., (2014), Gomez & Cook (2010), Arachchige et al., (2013) and Mattachini et al., (2017).

Conclusions

Literature values for time budgets of lactating dairy cows in the U.S. Midwest and South-Central Turkey suggest recoverable manure fractions of 80-100% for animals in freestall housing (no open lot access) and about 50% for those with access to open lots. Results from a recent study at four California dairies suggest 82-90% (87% range midpoint) and 43-49% (46% midpoint) recoverable manure fraction for lactating cows in freestall and open lot access housing, respectively. For dry cows, the California study suggests the manure recovery fraction ranges from 21-36% (29% midpoint).

Compared to the AMMP Quantification Methodology defaults, the recoverable manure fraction for lactating cows from the California literature is slightly higher for freestall housing (87 and80%, respectively) and likely significantly higher for open lot (46 and30%, respectively). For dry cows, the California based data appears to confirm the AMMP Quantification default of 30%.

Recommendations

No change to AMMP Quantification Methodology manure recovery defaults is recommended (more data is needed).

For dry cows, the measured time-on-concrete yields manure recovery close to the Quantification Methodology default. Besides, a moderate deviation from the Quantification Methodology default for this class of animal would have relatively small impact on the overall manure recovery value calculated by the Quantification Methodology because dry cows comprise only about 15% of the adult herd and excrete less than lactating cows due to lower feed intake.

While the time-on-concrete data for lactating cows suggests higher default recovery rates, there is not enough data (or dairies evaluated) to justify a change.

Because manure recovery rates play such an important part in the GHG inventory estimate, more time-on-concrete data collection is recommended from a variety of dairies (large, small, freestall with and without access to open lot).

Technology for monitoring health, productivity, and on-farm location of individual cows has advanced in the past 10 years. Activity sensors embedded in ear tags or ankle monitors, with appropriate router set-up, allow 24/7 monitoring (including location) of each cow on the dairy.[3] Further research should be explored that utilizes location-activity monitors to record and analyze data from a wide range of dairies in the State. Careful experimental design would be needed with sufficient justification that the dairies analyzed are representative.

[3] E.g., see Smartbow (https://www.smartbow.com/en/home.aspx), Cowmanager (https://www.cowmanager.com/en-us/Solution/Modules#findmycow), and others.

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2. Technical Review of Solids Separation Default Factors and Recommendations for Addition of New Factors

Summary of Available Data on the Relationship between Measurements of Total Solids and Volatile Solids

Background

Solids content of "as excreted" livestock manure (and the ratio of volatile solids (VS) to total solids (TS)) depends on species, animal performance, and specific dietary feed (ASABE, 2014). Average manure production and characteristics are commonly used for dairy facility planning. For dairy animals —including lactating cows, dry cows, and heifers—typical as-excreted TS is 13-15% and the ratio of volatile solids to total solids (VS:TS) is 84-86% (Table 1b of ASABE (2014)).[4] The MidWest Plan Service uses a volatile to total solids ratio of85% for all dairy animals larger than calves (Lorimor et al., 2004).

Solids content (both TS and VS) for collected or stored manure varies from farm to farm based on the type of housing, manure collection practice, bedding type, environmental conditions, and sampling location (Kirk et al., 2011). Bedding materials (composed of organic and inorganic materials), sand, and soil are collected with manure in flush, scrape and dry lot manure systems. The more inorganic material collected with manure, the lower the volatile to total solids ratio in the "bulk" manure mixture.

Volatile Solids and Total Solids Data

The volatile to total solids ratio for collected dairy manure in the literature varies from 35-89% (Table 2-1). For flushed manure at several California dairies measured at the inlet to a solid separator or as digester influent, the volatile to total solids ratio is 70% (± 8%) (Summers et al., 2013 a; Edalati et al., 2019).

[4] ASABE (2014) states these typical "as excreted" manure values are based on typical diets and animal performance levels in 2002 and that these estimates "become obsolete due to changes in animal genetics, performance potential, feeding program strategies, and available feeds."

Table 2-1. Volatile solids to total solids for collected dairy manure from the literature.

Sampled Manure Description	VS/TS (%)	Comments	Source
Flush (no separation)	59-78		Kirk et al., (2011); Edalati et al., (2019); Summers et al., (2013a)
Flush separator effluent	54-74		Edalati et al., (2019); Summers et al., (2013a)
Separated Solids	72-94		Edalati et al., (2019); Summers et al., (2013a)
Dry Lot-Scrape	39-69	Two measurements	Kirk et al., (2011)
Digester Effluent	49-75		Summers et al., (2013a)
Lagoon Water	48-52	Two measurements	Leytem et al., (2017a)
Lagoon Solids	27-64		Pettygrove et al., (2009)

Volatile Solids to Total Solids Ratio is Not Constant

While ‘as excreted’ manure properties are fairly consistent for dairy animals on similar feed, properties are highly variable once the manure enters the manure management system. As it moves through collection and into storage, the manure flow can pick up sand, soil, bedding material, and feed particles, and may be diluted with water (flush systems) that may be fresh from a well or mixed with recycled lagoon water with its own volatile and total solids content. Volatile and total solids in the liquid fraction of manure will change if it goes through settling or mechanical separation, or is stored long enough for evaporation or fermentation to occur. Because solids properties are so variable and dependent on numerous factors (including measurement site), total solids cannot be used as a proxy for volatile solids. Measuring TS and VS is not routine on dairies unless to confirm and check nutrient management plans, to prepare to install a digester, or audit a GHG reduction project. The measurement itself is straightforward for both TS and VS but usually requires an outside lab or service to conduct.[5]

Summary and Description of the Various Solids Separation Technologies

Most California dairies use some method of solids separation (30-40% use settling ponds or basins only and approximately 30% use mechanical separation, with or without settling basins), (Meyer et al., 2011). For mechanical separators, there is no definitive inventory but Table 2-2 lists mechanical separators in order from most to least common use in California (Meyer, 2019). Of these, drag-flight conveyor and two-stage sloped screen separators do not have default separation values in the AMMP Quantification Methodology.

[5] i.e., TS is the amount of sample left after heating to constant weight at 103 to 105°C (evaporation) and VS is the amount of TS material “lost on ignition” after heating to constant weight in air at 550°C (AWWA, 2017). See Lorimor et al., (2004) for testing recommendations.

Table 2-2. Mechanical separators used at California Dairies.

Separator Type	Relative Occurrence in California Dairies*	Included in AMMP Quantification Methodology
Sloped Screen	Most common	Yes
Two-stage Sloped Screen	Several, likely less than 10	No
Drag Flight Conveyor	Less common but significant	No
Rotary Drum Separator	One	Yes
Centrifuge	None	Yes
Screw press	Several, likely less than 10	Yes
Roller Press	None	Yes
Weeping Wall	Several, likely less than 10	Yes

Source: (Meyer, 2019)

Sloped Screen Separator

Sloped screen separators (also known as inclined or sidehill screens) use gravity to separate the liquid manure from the solids. Liquid manure is pumped to the top of the screen and flows down passing through the screen, while solids accumulate on and slide down the screen to a collection pad or auger (Figure 2-1). Some screens have smaller openings near the top and transition to slightly larger openings near the bottom, referred to as "hybrid screens". Screen separators have few or no moving parts and require little power. Some screen separator systems have a roller press at the lower end of the screen to further dewater separated solids. The roller press in this case does not remove additional solids since it only accesses the solids separated by the screen. Systems with a vibrating screen have an eccentric weighted motor and suspension system. Screens often come with a separate wash-down system to keep the screen from clogging. They also require a sump and pump to collect liquid manure and induce flow over the screen.

A

B

Source: http://usfarmsystems.com/

Figure 2-1. Sloped screen (A); and with discharge conveyor showing accumulated solids (B)

Drag Flight Conveyor Separator

A drag flight conveyor separator (also known as scraped screen) receives pumped manure at one end (or the lower end is submerged in the manure pit) and the solids are dragged along a perforated screen by paddles or “flights” and fall off the opposite end onto a collection pad or roller press (Figure 2-2). Drag flight separators have more moving parts than static or vibrating screens including chain, sprockets, drag flights, bushings, and an optional squeezer at the discharge.

Source: http://www.albersdairyequipment.com/manure.php & (Edalati et al., 2019)

Figure 2-2. Horizontal drag flight conveyor separator.

Rotary Drum Separator

A rotary drum separator contains a cylindrical screen section where separation occurs. A motor rotates the drum while liquid manure is fed into one end of the drum. The drum is lined with helix shaped internal flighting which moves the solids along the length of the drum as it rotates while liquid and small particles pass through the screen and are pumped (or flow) away (Figure 2-3). The solids exit the outlet end of the drum and are either processed further or conveyed to a storage location.

A B

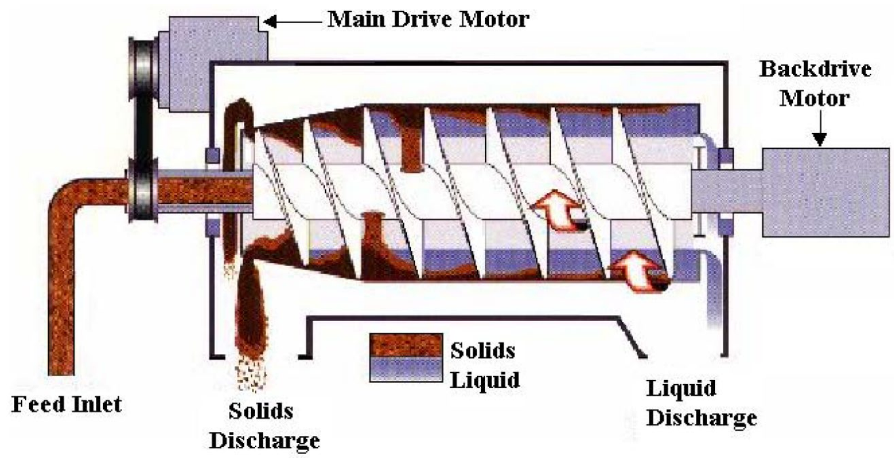
Source: Daritech http://www.daritech.com

Figure 2-3. (A) Rotary drum separator and discharge pile, and (B) internal view of the drum.

Centrifuge Separator

A centrifuge separator employs a high rotational speed to impart high centrifugal force in order to separate components of different densities. This reduces the settling time of the components to a matter of seconds which otherwise would take hours or days to settle. A decanter centrifuge (also known as solid bowl centrifuge) continuously separates solids from liquids in a manure slurry.

The slurry material enters the spinning centrifuge through a central inlet tube (Figure 2-4). The fast rotation generates centrifugal forces up to 4,000 g[6] (sometimes called "G-force") during which higher density components are collected and compacted on the inner wall of the bowl. A scroll (screw or screw conveyor) rotates inside the bowl at a slightly different speed and transports the settled particles along the wall to the solids discharge port. The clarified liquid (sometimes called centrate) flows out the opposite end.



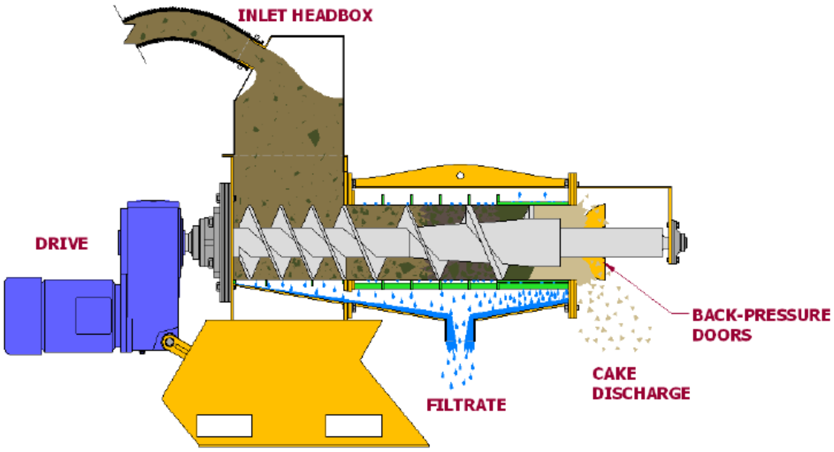
Source: Hutchison Hayes Separation Inc. www.hutch-hayes.com/en/

Figure 2-4. Centrifuge Schematic

Screw Press Separator

A screw press separator uses a cylindrical screen with a screw-type conveyor in the center. The screw conveys the solids retained on the screen while liquid and small particles drain (Figure 2-5). As the solids move along, the available volume decreases (the screw shaft diameter increases and/or the screw pitch is reduced) which creates increasing pressure forcing more water through the screen. Finally, the dewatered cake exits the end of the screw press through fixed openings or by pushing against a 'back pressure' door.

[6] g is the standard unit for the acceleration of gravity (32.2 feet/second^2, or 9.8 meters/second^2)



Source: Press Technology & Manufacturing: https://www.presstechnology.com

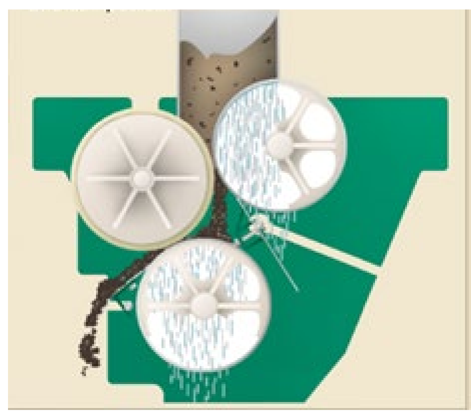
Figure 2-5. Screw press schematic

Roller Press Separator

A roller press separator forces material into a small space between a drive roller and one or more idler rollers. High pressure develops across a very small area pressing moisture out of the inflow material while the dewatered fibers continue through and are mechanically removed from the roller by either a scraper or a rotating brush (Figure 2-6).

A roller press works best with higher solids slurry manure (approximately more than 6% TS) or in series with a primary separator, such as a rotary drum or sloped screen. This device can used in place of screw presses where abrasion is a concern.

A. Roller Press Separator



B. Cutaway of roller press

Source: www.htequipment.net/Houle-Manure%20Separators.pdf

Figure 2-6. Roller Press Separator.

Weeping Wall Separator

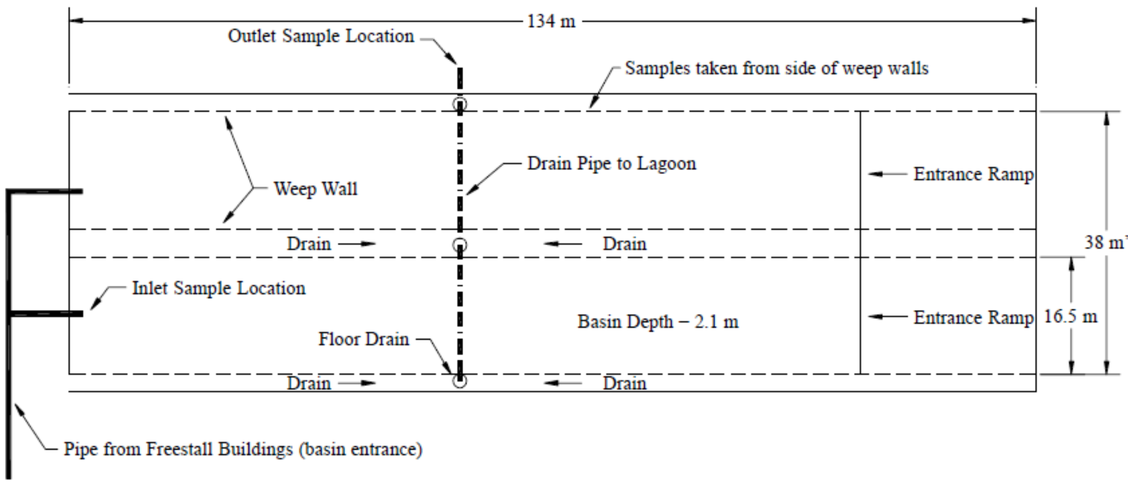
A weeping wall separator is a relatively large volume feature usually constructed with concrete with at least one porous wall that allows liquid and smaller particles to drain while solids are building up (Figure 2-7). Weeping wall systems are usually designed with two or more cells that are allowed to fill sequentially (Figure 2-8). While one cell is actively being filled, others are still draining or being cleaned by removing the leftover solids in preparation for a new filling cycle.

A. Adjacent cells; left is empty, right is filling

B. Liquid “weeping” through wall

Source: Edalati et al., (2019)

Figure 2-7. Weeping wall photos.



Source: Meyer et al., (2004)

Figure 2-8. Two-cell weeping wall settling basin schematic.

Settling Basins

A settling basin is used to separate and collect solids prior to sending the flushed manure to the storage pond or lagoon. It reduces solids load in the lagoon and increases its storage capacity. Settling basins need to have the accumulated silt or solids periodically removed (typically at least once per year) or they cease to be effective. A settling basin must be wide and long enough to reduce flow velocity (and turbulence) such that suspended particles have a chance to settle to the bottom before being carried out. Particle settling time is a function of its density, fluid viscosity, turbulence of the fluid, and total solids (Verley et al., 1974). Settling basins are best suited for low solids flows such as flush manure. Solids settling is hindered, or will take a very long time, for manures with TS > ~ 6% (Landry et al., 2004; Ackerman et al., 2010). For flushed dairy manure, 35-75% solids removal has been reported (Verley et al., 1974; Nye et al., 1976; Fulhage et al., 2002).

Settling basins are a significant source of methane emissions. Methane emissions measured in Idaho on dairies utilizing flush manure management indicated that settling basins emitted 40-50% of the total methane from the lagoon system, including storage lagoons and settling basins

(Leytem et al., 2017b). A similar study of two California dairies attributes 70-87% of liquid storage methane emissions to the settling basins (Arndt et al., 2018a).

Conclusions

Dairy manure solids separation efficiencies for several separator types were gathered from the literature and summarized in Table 2-3. Separators include sloped screen, two-stage sloped screen, drag flight conveyor, rotary drum, centrifuge, screw press, and roller press. Weeping wall settling basins are addressed in Task 3.

Summary of Available Data from Peer-Reviewed Literature and Other Sources on Separation Efficiency.

Sloped Screen

Sloped screen separation efficiency varies from 1-9% TS removal (Hegg et al., 1981) to 60.9% (Chastain et al., 2001). Hegg (1981) pumped low solids manure over a screen with 0.02-inch (0.5 mm) openings. Chastain (2001) measured solids removal using an Agpro separator with 0.06-inch (1.5 mm) openings. Summers et al. (2013) measured 19% and 40% TS removal and 33% and 42% VS removal at two California dairies as part of a dairy and digester mass flow and energy investigation. Screen opening sizes were not reported. Edalati et al. (2019) measured sloped screen separator efficiencies at two California flush dairies over multiple seasons during the year. TS and VS removal varied from 20- 49% and 26- 63%, respectively. The overall average was 34.9% and 44.7%, for TS and VS respectively. The screens were “hybrid” types, with smaller openings at the top (0.015-0.02 inch) transitioning to larger openings at the bottom (0.025-0.035 inch). Influent TS and VS varied from 1.2% to 2.6% and 0.8% to 1.9%, respectively. Zhang et al., (2003) measured solids in the influent and effluent of an inclined screen separator in ten sampling events over five months. The screen, with 2 mm (.08”) openings, received flushed manure with ~2.2% TS. Average removal efficiencies, based on concentration differences between influent and effluent, were 27% (range: 16.7- 40.6%) and 34% (range: 22.1- 47.9%) for TS and VS, respectively.

Two-Stage Sloped Screen

Edalati et al. (2019) and Chastain (2008) evaluated two-stage sloped screen separators, which were composed of two inclined screen separators operated in series (the first screen had 0.02-0.025 inch openings, and the second had 0.01- 0.015 inch openings). Both studies measured solids removal efficiencies of approximately 60 and 65% for TS and VS, respectively. For Edalati et al. (2019), influent TS and VS varied from 2.5- 3.7% and 2- 2.8%, respectively.

Drag Flight Conveyor or Scraped Screen

Edalati et al. (2019) also evaluated an horizontal scraped screen separator with 0.09 inch (2.4 mm) openings. Measured separation was only 8% and 12.1% for TS and VS respectively but input flow was larger than separator design capacity during these measurements. An unknown amount of excess flow bypassed the screen and mixed with the screen output liquid causing an inconclusive result.

Table 2-3. Separator efficiencies for dairy manure (Literature and AMMP Quantification Methodology values).

Manure Collection‡	Separator	Removal Efficiency (%)		Source
		Total Solids	Volatile Solids	
Flush	Sloped screen	**34.9** (20-49)†	**44.7** (26-63)†	Edalati et al. (2019)
Flush	Sloped screen	**27** (16.7-40.6)†	**34** (22.1-47.9)†	Zhang et al. (2003)
Flush	Sloped screen	60.9	62.8	Chastain et al. (2001)
Flush	Sloped screen	1-13	-	Hegg et al. (1981)
Flush	Sloped screen	19 and 40	33 and 42	Summers et al. (2013b)
Scrape	Sloped screen - screw press combo	25	-	Huijsmans et al. (1984)
Flush	Two-stage sloped screen	60	64.8	Edalati et al. (2019)
Flush	Two-stage sloped screen	59.7	65.7	Chastain (2008)
Flush	Drag flight conveyor*	8*	12.1*	Edalati et al. (2019)
Scrape	Centrifuge	62.7	-	Møller et al. (2007)
Scrape	Centrifuge	65.2	-	Møller et al. (2002)
Flush	Screw Press	30	29	Summers et al. (2013b)
Flush or Scrape	Screw Press	20-46	-	Hjorth et al. (2011)
Scrape	Screw Press	30	-	Møller et al. (2002)
Scrape	Screw Press	46-71	53-77	Gooch et al. (2005)
Scrape	Roller Separator	40	45	Gooch et al. (2005)
AMMP QM Separator Default Values				
-	Vibrating Screen		15	AMMP Quantification Methodology
-	Stationary Screen		17	
-	Screw Press		25	
-	Roller Drum		25	
-	Centrifuge		50	
-	Belt Press / Screen		50	
-	Weeping Wall		45	

Notes: ‡Flush manure has relatively low solids content (~ <5%). Scrape manure is consistent with a “slurry” with 6% < TS < 15%.

† **Bold** is average value. Measured range contained within (parenthesis)

*Inconclusive separation efficiency: Flow was larger than separator design capacity. Excess flow bypassed the screen.

Centrifuge

Møller (2002 & 2007) used a Pieralisi (Italy) decanter centrifuge rotating between 2000 and 4000 rpm delivering a separation acceleration of 2200–4100 g. Influent TS was 6- 7% and solids separation efficiency of 62.7- 65.2% was achieved.

Screw Press

Summers (2013) and Møller (2002) each reported TS separation from a screw press at 30%. Møller (2002) used a FAN screw press (FAN Separator GmbH, Marktschorgast, Germany) with screen opening diameter of 0.75–1 mm to remove solids from dairy manure slurry containing 7.1% TS. Summers (2013) did not describe the screw press tested. In a review article, Hjorth et al., (2011) lists several results from the literature with a range of 20-46% dairy manure TS removal efficiency. Gooch et al., (2005) evaluated separators on four dairies in New York. FAN screw presses with screen pores of 0.5–0.75 mm were used which yielded solids removal efficiencies of 46- 71% and 53- 77% for TS and VS, respectively. Influent manure solids content ranged from 7.5- 10 %.

Roller Separator

Gooch et al. (2005) also evaluated a Houle roller separator using 10% solids content manure as input and was able to separate 40% and 45% of the TS and VS, respectively.

Methane Production Potential and Solids Removal

Literature that report solids separation efficiencies and measured reduction in methane potential for dairy manure was also reviewed (Table 2-4 and Figure 2-9). As expected, the more VS removed from anaerobic storage, the lower the GHG potential.

Hills et al. (1985) investigated and compared methane production potential of untreated and 10 mesh filtered flushed dairy waste in 4 L laboratory digesters operated in continuous mode at 35 °C for 100 days. Untreated flush manure averaged 2.1% TS. Filtering through a 10-mesh laboratory filter removed 44% and 46% of the TS and VS, respectively. The filtered manure produced about 85% as much methane as the unfiltered manure (or approximately 15% methane reduction).

El-Mashad et al. (2010) filtered manure using a 2 mm screen and conducted assays of the untreated manure and the separated fractions using 1 L laboratory batch digesters; (at 35 °C, 30 days). The filtering removed 33.6% and 38% of the TS and VS, respectively. Based on methane production from the assays and relative mass fractions, methane production was reduced by 32% by separating solids.

Rico et al. (2007) evaluated methane production for solid and liquid fractions of dairy manure. Manure at 8% solids was collected followed by 1 mm screening of a portion of the manure. A calcium oxide (CaO) coagulant and a cationic polyacrylamide flocculent was added to the initial filtrate to further remove solids. Solids removal was 78% and 83% for TS and VS, respectively. Methane potential for raw manure and post screened and coagulated-flocculated liquid (filtrate) was determined using 2.5 L batch laboratory reactors operated at 35 °C for 45 days. Accounting for mass removed in the filtering step, the screened manure produced about 67% less methane than the untreated fraction. A follow-on lab study by Rico et al. (2012) separated dairy manure using a commercial polyacrylamide (Praestol K144L) and 1.5 mm and 0.2 mm screens, before and after flocculation, respectively. The results were removal of 77.5% of TS and 82% of the VS were removed. Methane production potential was reduced by 67%, the same result as Rico (2007).

Pain et al. (1984) operated two 125 m^3 mixed tank mesophilic digesters at a dairy, one fed with 7% TS dairy manure slurry and the other digester used the filtrate (4% TS) from roller press screen separator. The separator removed 49% and 52% of TS and VS, respectively. Accounting for mass removed in screen separator, the methane production from the filtrate was about 34% less than the raw manure slurry.

Table 2-4. Solids removal efficiencies and methane reduction potential from literature

Manure Collection	Separator	Solids removal (%)		Methane reduction (%)	Comment	Source (case)
		TS	VS	CH4		
Flush	Sloped screen	27.7	35.5	38.2		Edalati et al. (2019) (Farm A)
		42.9	51.9	43.3		
		44.8	58.4	57.2		
		48.9	62.6	61.2		
Flush	Two-stage sloped screen	60	64.8	66		" (Farm B)
		59.8	72.8	73.1		
		37.6	41.4	28.2		
Flush	Horizontal scraped screen	8	12.1	8.4	Overflow	" (Farm D)
Flush	Sloped screen	38.4	48.8	42.2		" (Farm F)
		20.1	26.4	28.9		
		21.8	29.5	37		
		31.7	41.3	38.1		
Flush	Advanced multi-stage	78.8	79.6	83.4		" (Farm C)
		65.1	66.9	71.9		
		64.2	62.7	69		
Scrape	Screw Press	75	78	80		VanderZaag et al. (2018)
Scrape	10-mesh screen	44	46	36	lab filtering	Hills (1985)
Scrape	2 mm screen	33.6	38	32	lab filtering	El-Mashad (2010)
Scrape	Roller Press	49	52	34	Digester	Pain (1984)
Scrape	1 mm screen & coagulant-flocculant	78	83	67	lab filtering	Rico (2007)
Scrape	1.5 & 0.2 mm screens & coagulant-flocculant	77.5	82	67	lab filtering	Rico (2012)

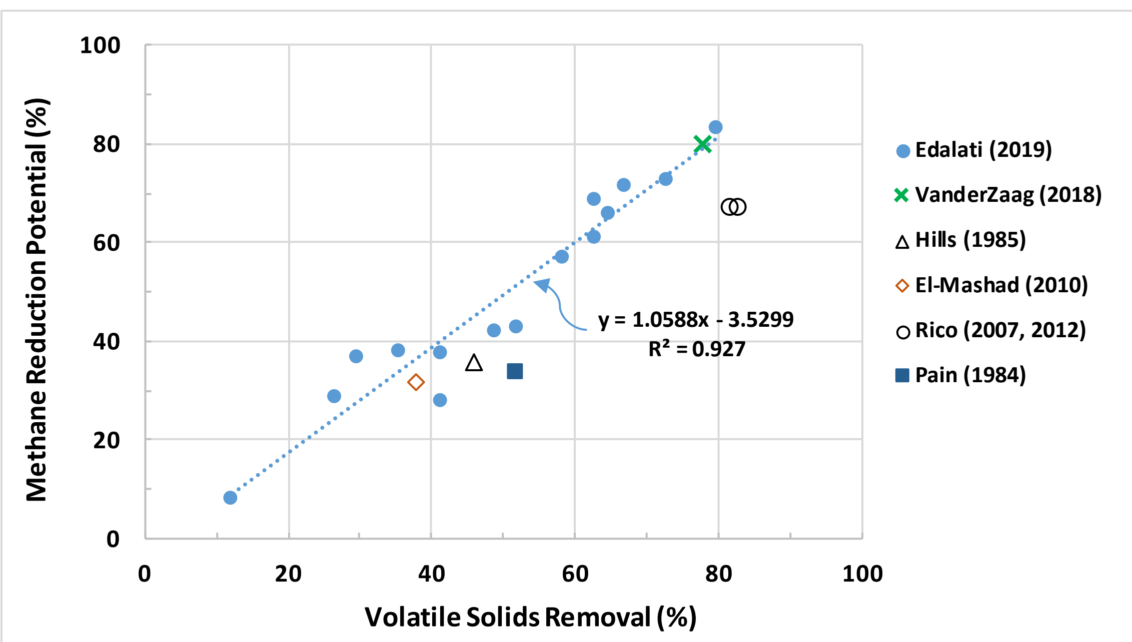


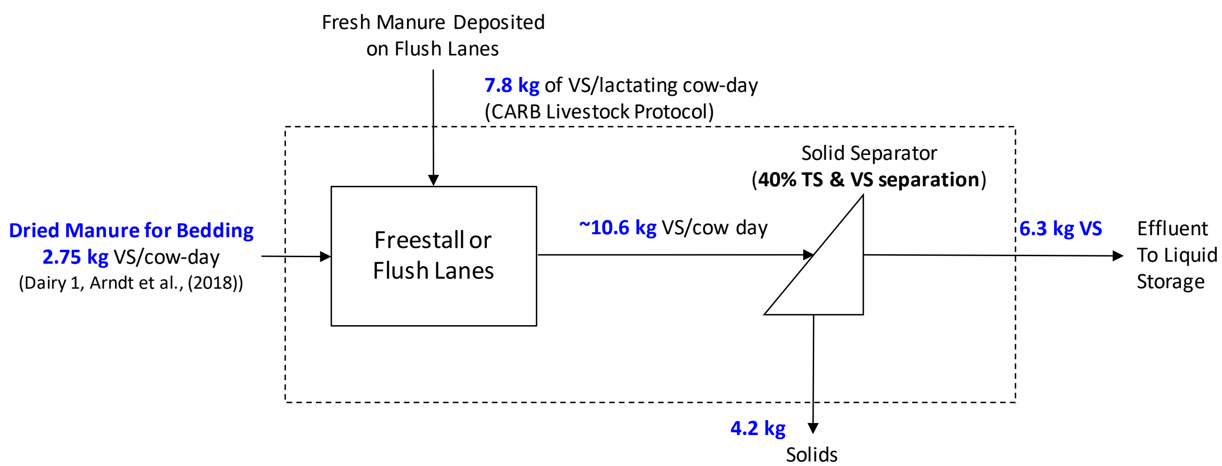
Figure 2-9. Methane reduction potential vs. volatile solids removal fraction from literature.

Effect of Bedding Material Volatile Solids on Greenhouse Gas Emissions from Stored Dairy Manure

Separated solids from flushed manure are commonly used as bedding on California dairies after being dried and/or composted (Carroll and Jasper, 1978; Meyer et al., 2019). Sometimes supplemented with nut shells, rice hulls, straw or other, recycled manure solids (RMS) are added as bedding to freestall housing from one to several times per week (Husfeldt et al., 2012). Some of this bedding material is flushed with manure adding to the total solids and volatile solids load sent to liquid storage. Neither the AMMP Quantification Methodology nor the Livestock Protocol, accounts for the impact of bedding material on GHG emissions estimates.

To improve GHG estimates from manure management, Meyer, et al. (2019) recommends investigating volatile solids flows (including bedding material) and fate (e.g., relative fractions that are decomposed anaerobically, aerobically, or are recalcitrant) on commercial farms. Le Riche et al. (2017) discovered that "bedding material used in dairy housing systems greatly influences subsequent GHG emissions from stored slurry". They monitored GHG emissions from separately stored slurry from a dairy in Canada that used sand bedding for one part of the herd, and wood shavings for the other. The slurry from wood shaving bedding produced 53% more CO_2-equivalent GHG emissions during the warm season than sand bedding slurry.

To account for the fate of bedding, Arndt et al. (2018) assumed that the amount of RMS bedding added per day or week (which averaged 2.75 kg of VS per cow per day)[7] was equal to the amount that was being flushed with manure. Based on that discussion, a mass balance across a simple freestall housing with a solid separator system can be analyzed (Figure 2-10). On a per cow, per day basis, 7.8 kg of VS from fresh manure is deposited in the freestall (or milking parlor – this assumes time on concrete is 100%) and flushed along with ~2.8 kg of VS from bedding (assuming steady state with no accumulation of bedding material) for a total of 10.6 kg VS as influent to the separator. For a "nominal" separator efficiency of 40%, 4.2 kg of VS are removed leaving 6.3 kg VS in the effluent sent to storage (Figure 2-10).



Note: This ignores any separable VS that comes in with flush water, which is typically drawn from lagoon storage.

Figure 2-10. Mass balance of volatile solids accounting for bedding; Barn-to-separator-to-liquid storage (Arndt et al., 2018 discussion).

[7] Meyer et al., (2019) reports bedding use as high as 9.5 kg VS per animal per day.

From this simple mass balance, an "effective" or apparent separator efficiency, using fresh manure input as the basis, is described in Equation 1:

$$= \left[1 - \frac{\text{Separator Effluent}}{\text{Fresh Manure Input}}\right]$$

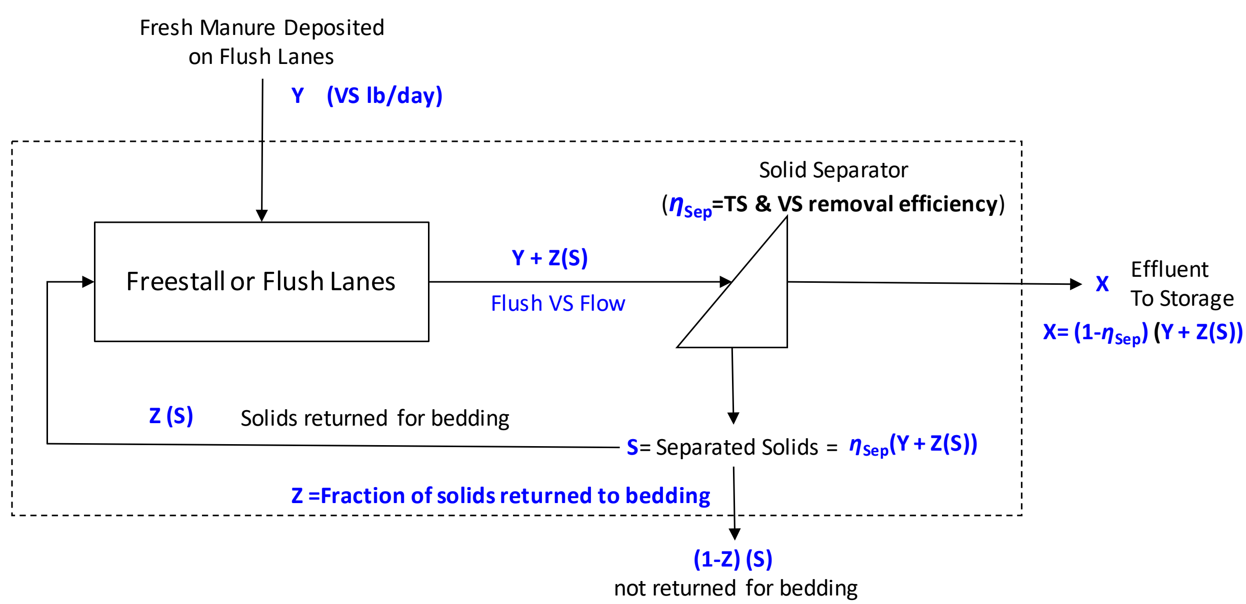
$$= \left[1 - \frac{6.3}{7.8}\right]$$

$$\approx 0.19 \quad (19\%)$$

[Equation 1]

For a separator with 40% solids removal efficiency (the "nominal" efficiency), the effective solids separation for the system, accounting for bedding material VS in the flush, is 19%.[8]

In the general case of RMS for bedding material with a solids separator with nominal efficiency of η_{Sep}, the fraction of solids recycled for bedding, Z, fresh manure entering the flush system, Y (lbs VS per day), and VS effluent to storage, X, is depicted in Figure 2-11



Note: This ignores any separable VS that comes in with flush water, which is typically drawn from lagoon storage.

Figure 2-11. Mass balance of volatile solids accounting for recycled manure solids for bedding.

Based on the above, Equation 2 depicts the effective or apparent separation efficiency of the system:

[8] If separable solids are drawn from the lagoon with the flush water, then solids load applied to the separator would be higher yielding more solids in the separator effluent. This also would contribute to a lower effective separator efficiency.

$$=\left[1-\frac{\text{Separator Effluent}}{\text{Fresh Manure Input}}\right]$$

$$=\left[1-\frac{X}{Y}\right],$$

[Equation 2]

Where X is separator effluent to storage and Y is fresh manure input to the flush lanes. With some simple algebraic substitution for X and Y and rearranging (not shown), the effective separation efficiency is a function of nominal separation efficiency (η_{Sep}), and Z (Equation 3).

$$=\left[1-(1-\eta_{Sep})\left(1+\frac{\eta_{Sep}{*}Z}{(1-\eta_{Sep}{*}Z)}\right)\right]$$

[Equation 3]

Where η_{Sep} is nominal separation efficiency and Z is the fraction of separated solids returned to bedding. Using equation 3, effective separation efficiency is plotted against fraction of recovered solids recycled for bedding for several "nominal" separation efficiencies in Figure 2-12.

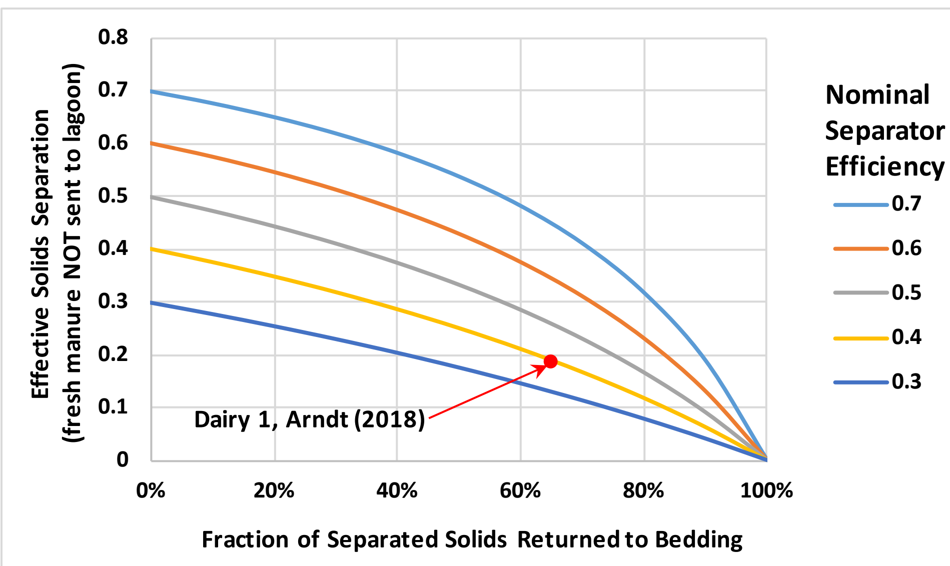


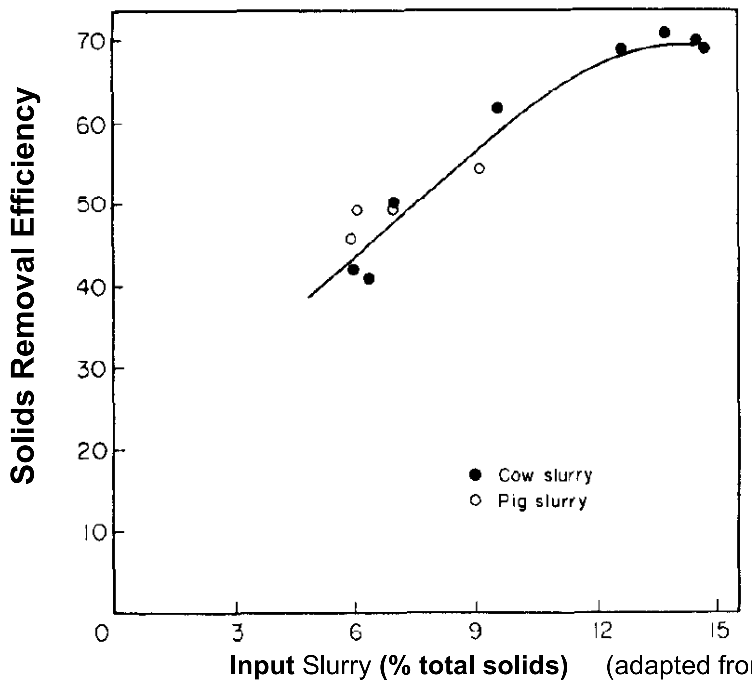
Figure 2-12. Effective separation versus fraction of solids returned to bedding.

With no accounting for bedding (or no bedding in the flushed flow), the "nominal" and effective separation efficiencies are the same. In the limit as RMS approaches 100%, the effective separation goes to zero and VS in the amount of daily fresh manure input flows to liquid storage. In a practical case therefore, bedding, spilled feed, and other solids flushed with fresh manure will cause more manure to pass through the screen and reach the storage lagoon and create methane emissions than otherwise would happen. Some of these adventitious solids (i.e., recycled manure bedding, straw, spilled feed) will contribute to manure emissions as well.

Recommendations for Solids Separation Factors in the AMMP Quantification Methodology

Stationary Screen

Solids separation efficiency for devices that incorporate screens, such as inclined stationary screens and screw presses, is generally a function of influent flow rate, screen opening size, solids particle size, fluid properties, and influent total solids concentration (Hjorth et al., 2011; Zhang et al., 2003; Hamilton et al., 2016). For example, Figure 2-13 displays solids removal efficiency vs. input slurry TS for cow and pig manure for a rotary screen separator (Pain et al., 1978).



(adapted from Pain et al., 1978)

Figure 2-13 Relationship between input total solids and solids removal efficiency for a rotary screen separator

In order to compare sloped-screen data in the literature, some of the variables that influence separation efficiency are approximately the same across those studies including the following:

- Influent flow rate: assuming the devices were operated at or near design flow
- Fluid properties: all studies involved flushed dairy cattle manure
- Solids particle size: though not reported, can assume particle size range is similar across the studies for flushed manure

With regard to opening size, some studies do not report size while those that do range from 0.02 to 0.08 inches. Influent TS concentration varies from 1% to 7% in the sloped screen literature. Assuming that opening size is similar across the studies, Figure 2-14 displays VS removal efficiency against the influent TS (also shown is a natural log curve fit of the data [orange dash line] and the 17% AMMP Quantification Methodology default separation value for stationary screens). A, B, and F are recent data from the University of California, Davis (UCD) study (Edalati, et al., 2019) with four seasons of data from Farm A, three seasons from Farm F and one measurement season from Farm B[9].

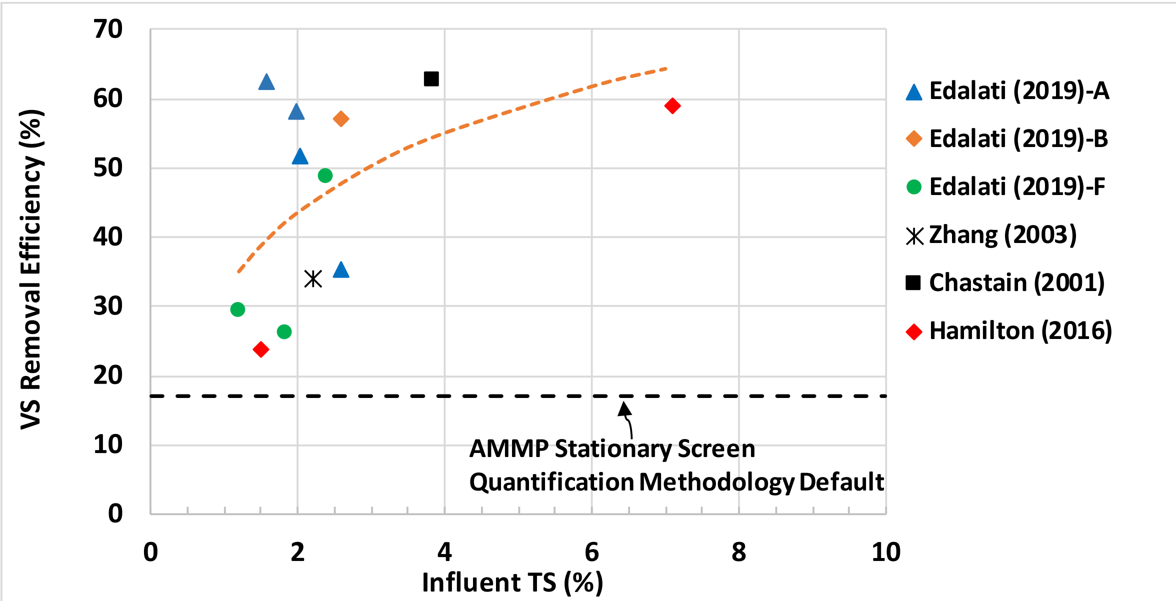
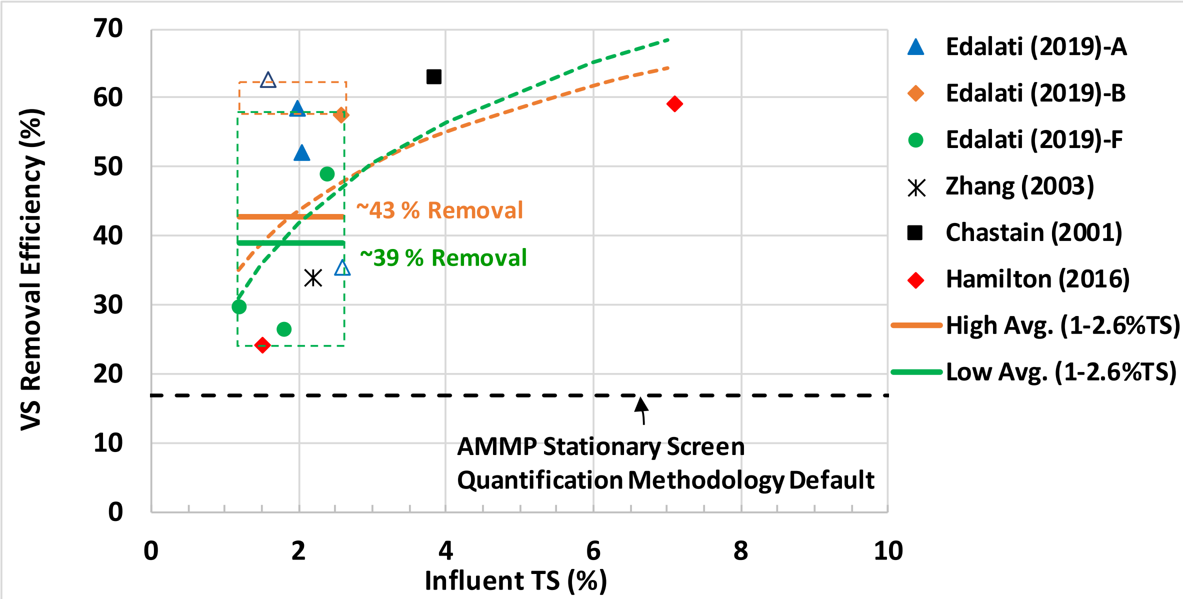


Figure 2-14. Volatile solids removal versus influent total solids concentration, sloped screen separators.

The data is generally consistent with increasing removal efficiency as influent solids concentration increases. However, the Edalati et al. (2019) Farm “A” data by itself seems inconsistent with this trend, where solids removal for the highest influent TS is the lowest of the “A” data set and removal for the lowest influent TS is the highest.

If the high and low of the Farm “A” dataset are removed, a slightly different trend line emerges (dashed green in Figure 2-15). Also shown in Figure 2-15 are dashed boxes incorporating data for typical flush manure influent solids (TS ≤ 2.6% for this data); the dash-green box for Farm “A” with high and low data removed, and the dashed orange plus green boxes incorporating all data for influent TS≤ 2.6%. The average removal efficiency for flush manure (dash-boxes) is 43% and 39%, for all data, and Farm "’A’ high and low exclusion," respectively.

[9] Showing data for only the first screen in a two-screen in series separator system at Farm B.



Notes: Orange dashed line is ln(x) curve fit of all data. Green dashed line is ln(x) fit of data after removing Edalati (2019)-A high and low. Dashed boxes indicate data for typical flush manure (influent TS ≤ 2.6% for this data). High average (43%) represents average of all data for influent TS ≤ 2.6%. Low average (39%) excludes the Edalati (2019)-A high and low.

Figure 2-15. Sloped screen volatile solids removal averages for low influent total solids concentration

These data warrant an increase in stationary screen default solids removal in the AMMP Quantification Methodology.[10] Given the large range in removal efficiencies listed in Table 2-3 (1% to 60.9% for TS and 22.1% to 62.8% for VS), a simple average of all data is not appropriate. However, for clearly flush manure cases, with the expectation that removal efficiency is lower for lower influent TS concentration (Pain et al., 1978; Hamilton et al., 2016), the data can be narrowed (i.e., dashed boxes in Figure 2-15) to an average of 39% to 43% for systems properly sized for the flow, and otherwise maintained and operated as designed.

Recommendation: Incorporate higher efficiencies indicated in the new data but also allow for off-spec operation (e.g., screen over flows, insufficient maintenance) and times of year with very low influent TS, a **30-35% default solids removal** for sloped (stationary) screen separators (Figure 2-16).

[10] If other VS flows and fates are accounted for and the Quantification Methodology is revised accordingly.

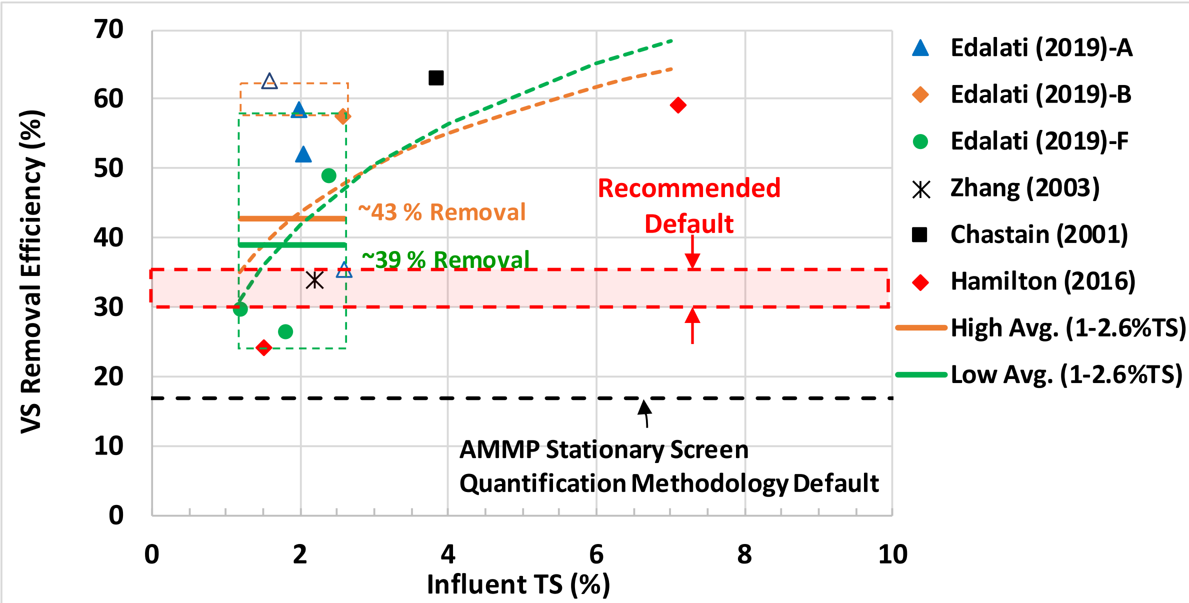


Figure 2-16. Recommended Quantification Methodology default for sloped screen separators

Two-Stage Stationary Screen (in series)

Both literature sources for two-stage screen separators (in series) measured essentially the same overall separation efficiency independently at about 65% VS removal (see Table 2-3).

Recommendation: Add a two-stage screen option (i.e., two stationary screen separators in series) to the AMMP Quantification Methodology for flush manure projects with a default solids removal of 50%. This allows for some off-spec operation and other non-ideal characteristics expected to occur with year round operation (as opposed to short term measured data).

Screw Press

Screw press separation efficiency versus influent TS concentration is plotted from two data sources in Figure 2-17. Burns & Moody (2001) tested efficiency of a Vincent KP-6L screw-press (Vincent Corporation, Tampa, FL) with 1.3 mm openings using dairy manure over a 1-10% influent TS range. Hamilton et al. (2016) reported removal efficiency for a screw press with 2.38 mm screen openings for dairy manure at several different influent TS concentrations. Both data sets display a strong correlation of removal efficiency with influent TS concentration.

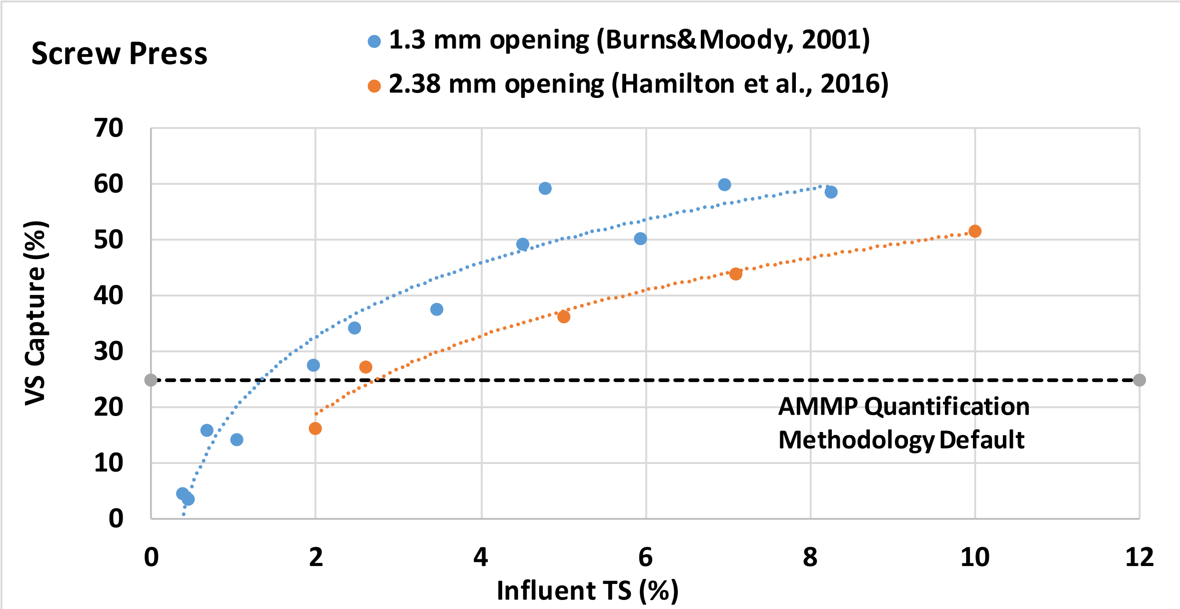


Figure 2-17. Volatile Solids removal versus Influent Total Solids Concentration; screw press.

Figure 2-18 displays the screw press data with a green dashed box enclosing low influent TS manure (representing flush manure systems), and a blue dashed box enclosing higher influent TS (representing consistency from scrape or vacuum collection). Removal efficiency data averages are 20% and 50% for the flush and scrape manure, respectively.

Recommendation: Based on this data, the recommendation is to keep the default removal efficiency at 25% for flush manure in a screw press, and add a category for scrape or vacuum manure collection systems using a screw press with a default of 50% for high TS slurry (i.e., > 6%).

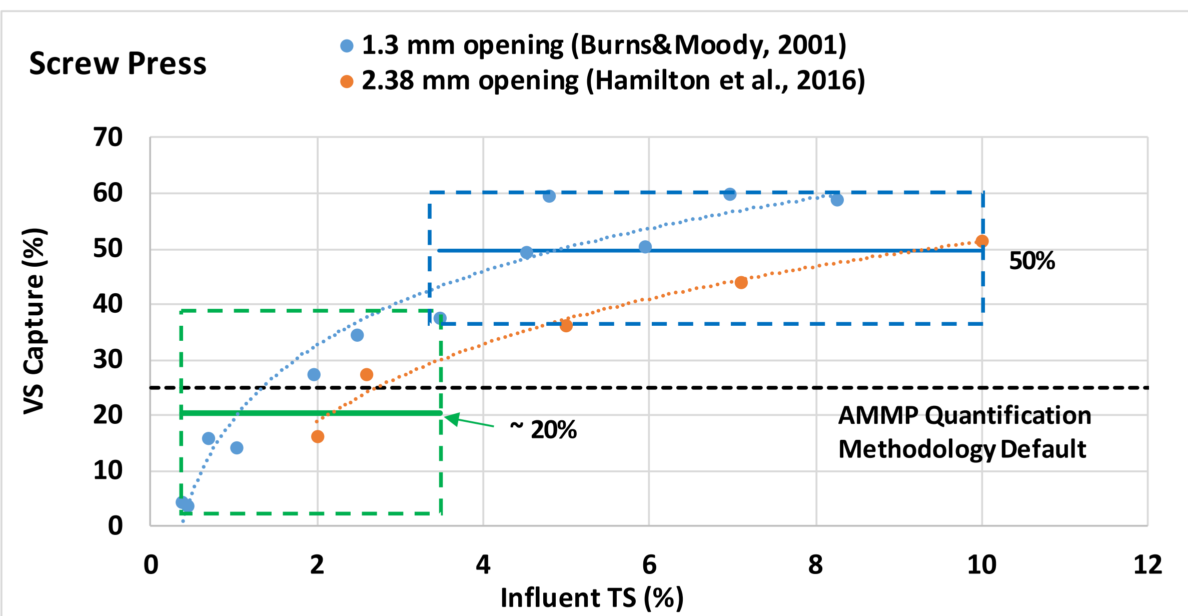


Figure 2-18. Screw press efficiencies; flush and slurry manure.

Drag Flight Conveyor/Scraped Screen

Though less common than the sloped screen solids separators, there is a significant number of drag flight screen conveyors (or scraped screen) separators used on California dairies (Meyer, 2019). The current AMMP Quantification Methodology does not address this type of separator so it is likely classified as a stationary screen by Quantification Methodology users. The Edalati et al. (2019) solids separator study includes one farm with a drag flight conveyor screen but it is undersized for the manure flow rate at that farm with significant overflow, or bypass, observed by the researchers. No other independent performance data were found for this separator type. Given that it is used in significant numbers in the State, and has a different principle of operation compared to a sloped screen, CARB should consider measuring the performance of two or three properly sized and operated drag flight conveyors in California and derive a separate Quantification Methodology default if results warrant.

Rotary Drum Separator, Roller Press, & Centrifuge

Recommendation: Given that there are few or none of these separators in use in California dairies and no separation data was found in the literature, no changes to the Quantification Methodology default for this type of device are recommended.

Weeping Wall

This is addressed in Chapter 3.

Summary Recommendations

Recommendations for "nominal" default separator efficiencies in the AMMP Quantification Methodology, reflecting newer information, are summarized in Table 2-5. In concurrence with Meyer, et al., (2019), it is also recommended that CARB evaluate the effect of other VS sources and flows, that are flushed with raw manure (e.g., bedding, feed, recycled manure solids), on effective separation efficiency and lagoon methane emissions. If deemed significant, these other VS flows should be accounted for in the Quantification Methodology.

Table 2-5. Summary of Recommendations

Separator Type	Recommendation	Current AMMP Quantification Method Default (Solids Removal)
Sloped screen	**30-35% default solids removal**	17%
Two-stage sloped screen	**50% default solids removal** (based on two studies with consistent results)	**None**
Drag flight conveyor	Conduct measurements on 2-3 drag flight conveyor separators in California and derive a QM default for this class of separator.	**None**
Rotary Drum Separator (Roller Drum)	No change	25%
Centrifuge	No change	50%
Screw press	**25% Default for Flush Systems** **50% Default for Scrape/Vacuum Systems**	25%
Roller Press (Belt Press)	No change	50%
Weeping Wall	Addressed in Chapter 3	45%

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3. Assessment of Methane Emissions from Weeping Wall Solids Separation Systems

Description of Weeping Wall System

A weeping wall system is an engineered settling basin with one or more perforated walls that form a large dewatering surface area (Meyer et al., 2004). Compared with mechanical separation technologies, a weeping wall offers several advantages including: lower energy consumption, repair, and maintenance cost (Mukhtar et al., 2011). They provide flexibility in manure management with extended storage periods (3-9 months) for manure solids. (Sustainable Conservation, 2005).

Generally, weeping wall systems consist of multiple rectangular cells (usually 2-4). Each cell is a standalone structure with concrete floor and one to three sides made of perforated walls (i.e., slotted concrete or screens supported by steel or concrete pillars (Mukhtar et al., 2011; Houlbrooke et al., 2011). The walls are typically 5- 8 feet tall (Meyer et al., 2004). The fourth side is used as an entry ramp for filling and emptying the cell.

Low solids flushed manure is directed to the basin entry and travels along the cell. Liquid drains through the weeping walls as solids accumulate inside the cell. The drained liquid usually flows to and is stored in lagoons for use as flush water and/or irrigation. The accumulated solids in the cell act as a filter that helps in capturing more solids. Depending on cell volume and number of animals, the cell can take 2-16 weeks to fill after which inlet flow is directed to the next empty cell. The full cell is left to continue to drain for up to ten more weeks after which the solids are removed with an excavator or a front loader and then composted or spread on nearby fields. The composted solids can be used as a soil amendment on the dairy farm or exported to other farms. Figure 3-1 shows weeping wall cells while they were empty, being filled and drained, completely filled, and filled and dried before emptying, respectively.

Figure 3-1. Weeping wall cells: empty with an entry ramp (top left), being filled and drained cell (top right), filled cell (bottom left), and filled and dried before emptying (bottom right)

Solids Removal Efficiency of Weeping Wall Systems

Results from recent work at UCD measured solids separation and recovery from two cells of a four-cell weeping wall system at an 8,000 cow dairy in California (Table 3-1) (Edalati et al., 2019). Average cell "filling" time (i.e., length of time flush manure was sent to the active cell) was 16 days followed by 4-6 weeks of draining and drying before solids were removed from the cell in preparation for another filling cycle. It took about 7 days to excavate a cell and recover the solids. TS removal from effluent flowing to storage lagoons (solids removal efficiency) averaged 80%. Meyer et al. (2004) evaluated a weeping wall at an 1,100 cow commercial dairy in California by sampling manure flows during three events in March and once in July. The influent mean TS concentrations was 1.5%. The weeping wall removed manure particles greater than 0.125 mm. The average TS removal ranged from 50-60%. The U.S. Department of Agriculture Natural Resources Conservation Service recommends solids removal value of 50-85% for weeping wall systems when doing preliminary facility design calculations (NRCS, 2014). The vendor for the "Tri-Bar weep wall" system claims effective solids removal of 60- 85% and up to 70% of sand (Nooyen, 2018). Finally, Mukhtar et al., (2011) evaluated a two-stage weeping wall system in east central Texas. The system was comprised of primary and secondary weeping wall stages. The effluent from the primary chambers flowed to the secondary chambers. Storage capacity in the primary and secondary systems was 60-90 days, and 21 days, respectively. Overall solids removal efficiency was 88%.[11]

[11] There are no known "two-stage" weeping wall systems in California.

Table 3-1 Solids removal efficiencies for weeping wall systems

Solid Removal Efficiency (%)	Source
78 - 82	Edalati et al. (2019)
50-60	Meyer et al. (2004)
50-85	NRCS (2014)[1]
60-85	Nooyen (2018)[2]
88 (two-stage system)	Mukhtar et al. (2011)[3]

Notes: 1. Recommendation for use in facility design. 2. Vendor claim. 3. Two-stage weeping wall system.

Emissions from Weeping Wall Systems

There is little published data on the emissions of different gases from weeping wall systems. Phillips et al. (1997)[12], cited by Chadwick et al. (1999), measured negligible N_2O emissions from a weeping wall that stores cattle slurry. The authors attributed this to the low nitrification rates due to the low oxygen concentrations in the weeping wall cells, but methane was not well characterized.

Van der Weerden et al. (2014) measured the emissions of N_2O, methane and ammonia (NH_3) from dairy manure storage (including in a weeping wall cell) and land-application in southern New Zealand. The manure stored in the weeping wall system came from daily scraping of the concrete feed lanes from which an estimated 60% of the liquid eventually seeped into a storage pond during the time manure was in the weeping wall cell. TS of the retained solids was 12%. The emissions (shown as percent of initial total nitrogen or carbon) for a weeping wall compared with deep litter for storage periods of 12, 16, and 28 weeks are shown in Table 3-2. The 12-week storage period occurred mid-winter through mid-spring when manure temperature averaged 11 °C. The next 4 weeks (until the end of the 16 week storage period) occurred in the spring with average manure temperature of 14 °C. To accumulate 28 weeks total storage time, the manure and emissions were monitored through the summer during which time manure temperature averaged 20 °C. For all storage times, cumulative methane emissions were significantly greater from the weeping wall manure while N_2O emissions were lower (compared to the deep litter manure storage). The low emissions of N_2O from the weeping wall solids were attributed to high moisture contents (88%, wet basis) that reduces the rate of nitrification and/or denitrifying. The higher emissions of methane from the weeping wall solids were attributed to predominantly anoxic conditions of the cell.

Table 3-2 Cumulative ammonia, nitrous oxide, and methane emissions (% of initial total nitrogen or carbon) from different manures

Storage time (weeks) / Storage type	Ammonia			Nitrous oxide			Methane		
	12	16	28	12	16	28	12	16	28
Weeping wall	5.35	5.67	5.67	0.01	0.09	0.20	0.21	0.27	5.10
Deep litter	1.15	1.16	1.16	1.93	1.93	1.93	0.01	0.02	0.74

Source: van der Weerden et al. (2014)

[12] Phillips et al. (1997) concluded methane emissions measurements were inadequate (problem with measurement technique).

The effect of temperature on methane emission rate is apparent in van der Weerden's data where 95% of methane emissions occurred during the final 12 weeks of storage (the summer months) when manure temperature averaged the highest (20 °C compared to 13 °C for the first 16 weeks). Furthermore, the summertime methane emissions (5.1-0.27% = 4.8% of initial carbon in the manure) is equivalent to 20% of the total methane producing potential (B_o) for dairy cow manure in the CARB AMMP Quantification Methodology (i.e., a MCF of 0.2).[13]

Average summertime temperature in the San Joaquin Valley varies from 35 °C to 16 °C, between daytime high and nighttime low, respectively (NWS 2019), suggesting larger methane emissions potential from weeping wall systems in California than identified by van der Weerden in New Zealand.

Anaerobic "Stacked Manure" Emissions:

While no other studies of emissions from weeping wall systems were found in the literature, there are a number of investigations that evaluated moist solid (nearly slurry) manure in anaerobic heaps or "stacks" on concrete surfaces or in concrete bunkers with liquid drainage (Table 3-3). These studies were of dairy or beef cattle manure at 15-20% initial TS. As such, these may be analogs to weeping wall systems towards the end of their drying and storage period (based on moisture content of weeping wall cells discussed in the following section). However, the emissions during the filling and initial drying phases of a weeping wall may be significantly different from stacked manure solids. A set of phase-specific MCFs (i.e., filling, drying/storage, and clean-out) during the weeping wall operation cycle might be the best approach to model emissions (see Figure 3-4 below).

Amon et al. (2001) measured methane emissions from both "anaerobically stacked" and actively composted dairy farmyard manure to determine emission factors for the Alpine regions of Austria, Switzerland and Southern Germany. Approximately 7 tons of manure (at 20% TS) was arranged in a "stack" on a concrete floor to fit underneath an open-ended emissions measurement chamber that covered a 27 m^2 (9 x 3 m) floor space. The height of the manure stack was about 0.6 m.[14] The stack was not constrained and any liquid seepage flowed outside of the coverage area. The storage period was 80 days with the experiment carried out twice, once during summer and once during winter where mean manure temperatures were 35 and 22 °C, respectively. MCFs for winter and summer were measured as 1.6% and 4%, respectively. This study serves as the basis for solid storage MCF values used in the IPCC Tier 2 guidelines (IPCC, 2006; IPCC, 2019).

Chadwick (2005) measured the effect of compaction during storage of beef cattle manure on emissions. Storage periods of 90, 96 and 109 days were used, two of which occurred during summer months. For each replication, approximately 2 tons of manure at 20% TS was stored in

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[13] The average carbon content of dairy manure, on a dry, ash-free basis (i.e., VS) is 51% by mass (Li et al., 2013 &; Fernandez-Lopez et al., 2015). van der Weerden's summertime methane emission of 4.8% of initial carbon in the manure is: 0.048 x .51 x 16/12 = 0.0326 kg CH_4/kg VS, where 16/12 is the ratio of molecular weights for CH_4:C. The B_o for dairy manure is 0.24 m^3 methane/kg VS (CARB AMMP Quantification Methodolgoy). Converting B_o from volume to mass: 0.24 m^3 methane/kg VS x 0.68 kg methane/m^3 methane = 0.163 kg methane/kg VS. MCF = actual emissions / B_o = 0.0326/0.163 = 0.2.

[14] Based on the area of the chamber and the density of manure at 20% TS (~0.8 tonne/m3, [Wang et al, (2019), Landry et al, (2004), and Houlbrooke et al, (2011)]).

concrete bunkers with grated floor drains. Methane emissions ranged from 0.4% to 9.7% of total carbon in the manure, which corresponds to MCF of 0.02 to 0.4, respectively.

Hu et al. (2018) measured the effects of temperature and moisture content on carbon dioxide and methane fluxes from fresh dairy manure as it dried over a 15 day period. One kg samples of homogenized, 15% TS, fresh dairy manure were placed in 203 mm diameter open containers and placed under soil gas flux measurement chambers. Sample depth, or "thickness" was about 30 mm (~ 1 ¼ inches). After 15 days, moisture decreased from 85% to 63% where methane emissions were no longer detected. Peak methane flux (emission rate) occurred at 80% moisture content while peak carbon dioxide emissions rate occurred at 75% moisture. Peak methane is expected at higher moisture content but gas diffusion in liquid is limiting and with small samples, no internal liquid circulation occurred and the methane production rate was too slow to create gas bubbles. As the sample dried, aerobic processes began to overtake methanogenesis leading to the carbon dioxide peak. Total methane emission was 0.05% of initial carbon content, which is equivalent to an MCF of 0.002.

Manure stored at higher moisture content has increased methane emissions, reaching a maximum as moisture content approaches 80%, which is the borderline between solid and liquid/slurry manure in the IPCC guidelines (i.e., liquid/slurry manure is ≤ 20% TS) (IPCC, 2019).

Table 3-3 Methane conversion factor values for a weeping wall system and solid manure storage from literature

Percent of Carbon converted to Methane	MCF*	Description	Source
5.1	0.21	Weeping Wall, NZ (7 months)	van der Weerden (2014)
4.8	0.20	Weeping Wall, NZ (3 warm months)	van der Weerden (2014)
9.7	0.40	Compacted solid manure "anaerobic" stack (3 months- Summer)	Chadwick (2005)
0.05	0.002	30 mm deep, 1 kg samples (15 day drying time)	Hu (2018)
3.5	0.15	Average of solid manure stacks in literature	Webb (2012)
0.96	0.04	Anaerobic stacks -Alpine summer conditions (3 months & basis for IPCC Tier 2 inventory guidelines)	Amon (2001)
-	0.27 – 0.46	Deep bedding manure (for average annual temperature of 15 and 21 °C, respectively)	CARB 2014

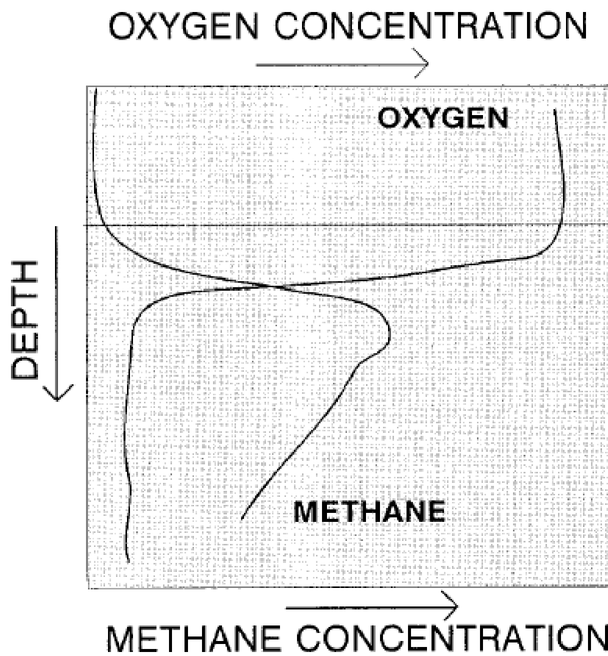
*Based on manure VS carbon content = 51% by mass, B_o = 0.24 m^3 methane/kgVS and 0.68 kg CH_4/m^3

Anaerobic Conditions are Present in Solids Accumulating in a Weeping Wall System

Once manure is excreted, biological decomposition and gas formation continue. Temperature and moisture content are the primary environmental factors that influence gas emission rates, as they directly impact nutrient availability, metabolic activity of microorganisms, and gas diffusion (Hu et al., 2018). High moisture content correlates with anaerobic conditions in porous media like soils and stacked manure solids because the pore space is filled with water (the water filled pore space is high). From soil science literature, for water filled pore space values greater than approximately 90%, anaerobic decomposition dominates. As the matrix dries, aerobic decomposition also begins to occur and the combination of anaerobic plus aerobic degradation rate reaches a maximum generally around 60% water filled pore space (Fichtner et al., 2019). The relationship between water filled pore space and gravimetric moisture content depends on porosity and bulk density of the dry substrate.

In addition to decomposition of organic solids to carbon dioxide and water during aerobic biodegradation, methane can be oxidized by methanotrophic bacteria present in the substrate. In terrestrial and wetland ecosystems, methanogens and methanotrophs are active simultaneously (methane generation and oxidation occurs simultaneously). Depending on conditions, such as moisture or water filled pore space, soil can be a methane source or sink.

In moisture-stratified systems such as peat bogs, landfills, and manure solids in a weeping wall cell (where drying is occurring at the top surface exposed to air), methane generation is at a local maximum at the oxic-anoxic interface where oxygen diffuses from above (Figure 3-2). Methane is produced in the lower oxygen-depleted layer. As drying occurs from above, pore space increases and air diffuses into media. Methanotrophs at the interface oxidize methane as it diffuses upwards. The amount of methane oxidized (and the amount that is emitted) depends on depth of the oxygenated zone, porosity, and gas diffusion properties. The scale of the depth can vary from several centimeters as in the case of a peat bog, to a few millimeters as in the case of an aquatic sediment (Topp & Pattey, 1997).



(Source: Topp & Pattey, 1997)

Figure 3-2 A representation of an oxic-anoxic interface in a vertically stratified system

During excavation of a drained weeping wall cell that had accumulated solids for 12 weeks, Meyer et al. (2004) measured TS and VS at six locations along the axis of the cell (from entry ramp to opposite end). Moisture and VS/TS were relatively low near the entrance where large amounts of sand were observed (Figure 3-3). Further from the entrance, the amount of sand was much less while moisture and VS/TS increased to 86% and 80%, respectively (Figure 3-3).

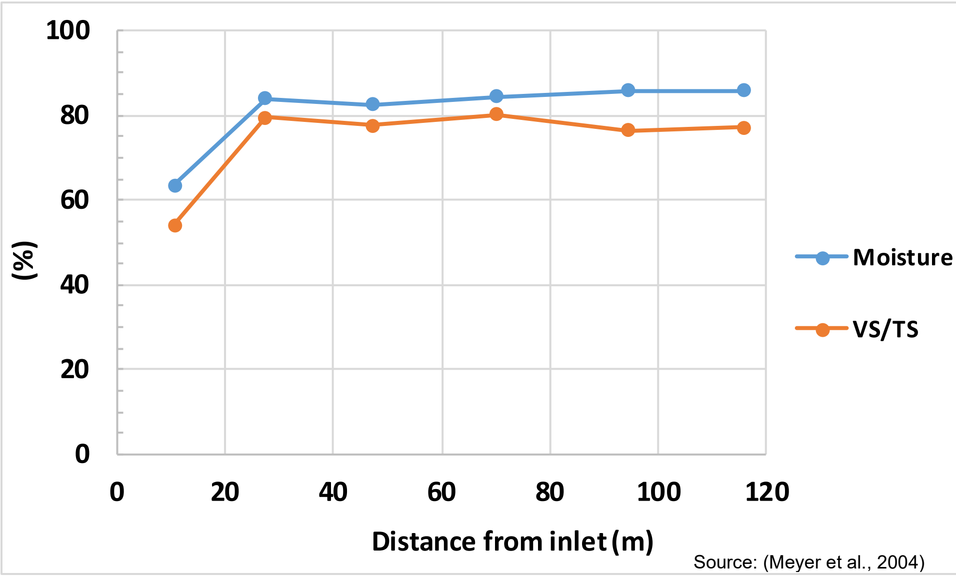


Figure 3-3 Moisture and volatile solids/total solids versus distance from inlet in a dewatered weeping wall cell

Results from a recent study at UCD that monitored solids separation efficiency in two weeping wall cells measured TS of excavated solids of 22% and 26% (78% and 74% moisture) (Edalati et al., 2019). Additionally, van der Weerden et al. (2014) measured 85% moisture content of excavated weeping wall solids.

The relatively high moisture contents measured in drained and excavated weeping wall solids reported in these studies indicate little or no air has penetrated deep into the cell mass and one can conclude that anaerobic conditions exist through much of the cell depth. Absent direct emissions measurements of weeping wall systems under California conditions, both the vertical moisture profile (and/or depth of air diffusion over time) and a better understanding of methane oxidation in manure crusts are needed to estimate the net methane emissions of a weeping wall cell throughout its fill, drain and excavation cycle.

Proxy for Weeping Wall Methane Conversion Factor for Use in the AMMP Quantification Methodology

The methane emissions from a weeping wall cell, measured by van der Weerden et al. (2014) during the summer in New Zealand (when manure temperature was ~20 °C), were equivalent to a 20% MCF. Emissions from "anaerobic stacked" manure range from MCF = 0.04 (summer Alpine conditions) to MCF = 0.40 (3 summer months compacted stacked manure in the UK), with a literature average of MCF =0.15 (Table 3). The MCF for deep bedding systems and liquid/slurry storage in the San Joaquin Valley[15] are 0.32 and 0.2, respectively (CARB Compliance Offset Protocol Livestock Projects, Table A.5 (CARB, 2014)).

Solids stored in a weeping wall cell may behave similar to "anaerobic stacked" or a deep bedding manure system while during the fill period the material is perhaps akin to slurry storage with crust cover.

[15] Average annual temperature of 17 °C (NWS 2019)

A Simple Model for Weeping Wall System Emissions

To model weeping wall system emissions, a set of phase-specific MCFs during the operation cycle (i.e., filling, drying/storage, and excavation) is proposed (Figure 3-4). For the initial fill period (which might take between 2-10 weeks), the material in the weeping wall basin will average very high moisture (i.e., less than raw flush manure (approximately 97-98% moisture) but more than during the storage and excavation phase (approximately 85% moisture)). Additionally, moisture content would be somewhat stratified with the top few inches of material exposed to drying developing oxygen infused pore space. Therefore, the fill state is assumed to mimic 'liquid/slurry with a crust cover' storage with respect to methane emissions. Assuming a constant rate of filling during the period, the fill period MCF would be half of the liquid/slurry with crust cover MCF for a full basin (or 0.10).[16] For the storage/drying period, the full deep bedding manure MCF (0.32) applies; for the excavation period, assuming more or less constant rate of removal, half deep bedding manure MCF (0.16) is used.[17]

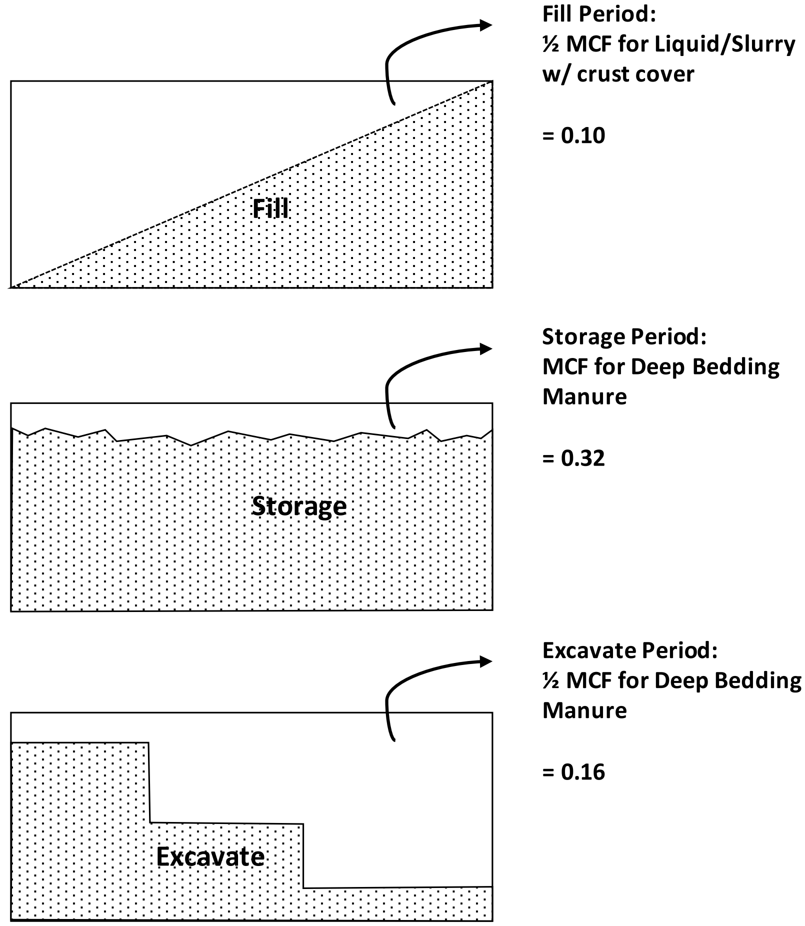


Figure 3-4.Representation of effective methane conversion factor fill, storage, and excavate phases of the weeping wall cycle

[16] Use half liquid/slurry with cover MCF because the basin averages half-full from start of fill to finish (assuming linear fill rate).
[17] Again, assuming a linear excavation rate, use half full-basin MCF for this stage.

Assuming each weeping wall cell is continuously cycled through fill, storage, and excavate stages throughout the year, an average weeping wall MCF can be calculated based on the relative time spent in each stage.

Using the limited data reported for California from Edalati (2019) and Meyer (2004), average time for weeping wall cell fill, storage, and excavation is 43, 49, and 7 days, respectively (Table 3- 4). The corresponding cycle time fractions are 0.388, 0.527, and 0.085, respectively.

Table 3-4 Weeping wall cell fill, storage, and excavate durations

	Days			**Fraction of cycle time**		
	Fill	**Storage**	**Excavate**	**Fill**	**Storage**	**Excavate**
Edalati (2019)	16	35	7	0.276	0.603	0.121
Meyer (2004)	70	63	7	0.500	0.450	0.050
Average	43	49	7	0.388	0.527	0.085

Multiplying the individual stage MCFs by their respective fraction of cycle time gives the individual stage MCF contributions, and summing yields, the total weeping wall MCF is 0.22 for solids retained (Table 3-5).

Table 3-5 Effective weeping wall methane conversion factor

	Fill	**Storage**	**Excavation**	**Total**
Stage MCF	0.1	0.32	0.16	-
Fraction of cycle time	0.388	0.527	0.085	1
Effective MCF	0.039	0.169	0.014	**0.22**

The overall system (or dairy) MCF would be the sum of effective weeping wall MCF and effective lagoon MCF (for the material *not* retained in the weeping wall cell). With no weeping wall, the system MCF is equivalent to 100% flush manure to lagoon (about 0.76 for dairies in the San Joaquin Valley). For a weeping wall that theoretically captures 100% of flushed manure solids, the MCF overall would be 0.22 (Figure 3- 5; solids capture fraction 1 = `100 %).[18] Using the recommended weeping wall solids capture of 65% (Table 3-1), the effective weeping wall MCF would be 0.144 (65% x 0.22). The overall MCF (lagoon and weeping wall) is 0.41. In this model. Therefore, retention of 65% of solids in the weeping wall reduced overall methane emissions by 46% ((1-.41/.76) x 100%).

[18] If the fill stage is represented as half MCF of liquid/slurry storage *without* crust cover (half MCF = 0.16) then the weeping wall effective MCF would be 0.24, all else equal. If the fill stage is represented as half MCF of anaerobic lagoon (half MCF = 0.38), then the effective weeping wall MCF would be 0.33.

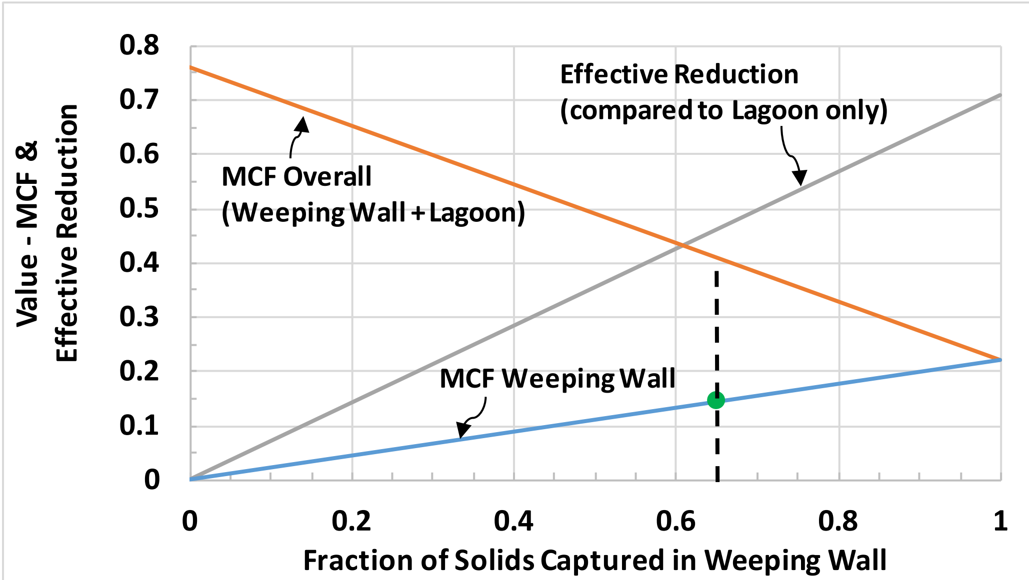


Figure 3-5 Effective weeping wall and overall methane conversion factors versus fraction of solids captured.

While this model is quite simple, it represents a first-cut approach to estimating weeping wall GHG emissions based on a small amount of literature and existing MCF values for storage systems that are analogous to discreet phases of a weeping wall system operational cycle. These four bullets discuss potential drawbacks to the model or areas that a more rigorous model might include:

- The fill stage, modeled relatively conservatively as "liquid/slurry storage with natural crust cover" (MCF = 0.2, half MCF = 0.1), could also have been modeled as "liquid/slurry without cover" (MCF = 0.32) or even "anaerobic lagoon" (MCF = 0.76), both with more liberal MCF values that would contribute to a larger effective MCF.
- In a real system, there is a lag phase of several days before methanogenesis is established (El-Mashad et al., 2003), or perhaps longer, until sufficient depth is obtained for anaerobic conditions (approximately 1 m for anaerobic lagoons), which would yield a lower effective MCF.
- During the second, or "storage" stage, liquid drainage could leach or remove some of the fermentation/digestion intermediates (e.g., volatile fatty acids) which would tend to reduce methane production relative to the "deep bedding" analogy where no leachate leaves the storage system.
- Finally, this model applies stage-specific MCF values to the total non-degraded VS captured by the weeping wall system. A more complex model would account for VS degraded by each phase leaving a smaller amount of VS available for degradation in the following phase. For this simple model, accounting for VS degradation by stage reduces the effective MCF by ~ 5% (0.21 rather than 0.22).

Conclusion

Weeping wall systems undoubtedly experience anaerobic conditions for much, if not all, of their cycle (fill, drain/dry, and excavate). Significant methane emissions from a weeping wall system and anaerobic stacked manure (an analog to weeping wall storage) have been measured in cooler climates than the San Joaquin Valley (i.e., New Zealand and the UK).

However, while weeping wall systems do remove 50-80% of solids from liquid manure flow to a storage lagoon (and reducing methane emissions from the lagoon), they do emit methane during the months-long "separation and solids storage" interval before excavation to composting. A simple model was developed and is proposed that includes stage-specific MCFs and relative stage duration to estimate an effective weeping wall system MCF of 0.22 for retained solids.

Recommendations

Recommend a 65% solids retention default with a MCF of 0.22 for weeping wall systems in the Quantification Methodology (the current default is 45% solids retention with no methane emissions (0 MCF)). If adding the 0.22 MCF, or similar, to the Quantification Methodology is not acceptable, then no change to the weeping wall solids retention default is recommended.

In addition, a weeping wall measurement and modelling research program is recommended to obtain a more complete understanding of the system emissions.

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4. Assessment of Commercially Available Aerobic Treatment Systems, and Recommendations Regarding Research Needs to Evaluate an Aerobic Treatment System

Background

Aeration is the process of mixing air (or oxygen) into liquid manure during storage to facilitate growth of aerobic bacteria. Under aerobic conditions, organic matter is degraded into carbon dioxide, water, ammonium (or nitrate), and phosphate, with a small accompanying amount of cell growth. Aeration techniques have been used at municipal wastewater treatment facilities for years to stabilize (oxidize) waste and provide odor control, and in swine manure management for similar purposes.

Lagoons can be aerated naturally by air/oxygen diffusion at the lagoon surface and/or by oxygen produced from the photosynthesis of algae. A naturally aerated lagoon is necessarily shallow, with a depth less than approximately 1 meter. A naturally aerated lagoon would require a larger area of land in order to have the same volume as a deeper anaerobic lagoon. Lagoons also can be mechanically aerated using compressed air bubblers, mechanical surface or subsurface mixers, pumped liquid-air-ejector systems, or some combination (Cumby, 1987).

Aeration has been used for odor control for many years, primarily in swine manure storage lagoons (Zhang & Zhu; 2005). Odor is reduced due to the aerobic biological degradation of the volatile fatty acids (VFA) and other compounds in manure. Methane and ammonia emissions can be reduced via aeration, though there is a potential to increase N_2O emissions (Loyon et al., 2007; Molodovskaya et al., 2008).

Literature Review

There are relatively few references for performance of aerated dairy lagoons or liquid dairy manure compared to literature on the aerobic treatment of swine manure.

Dairy Manure

Sukias et al. (2003) investigated the effectiveness of adding mechanical aeration to existing facultative (shallow, partial aerobic) lagoons at a dairy in New Zealand. The continuously aerated lagoon removed 80% of 5-day biochemical oxygen demand (BOD_5)[19] compared to 67% for the non-aerated facultative lagoon. Ammoniacal-N removal was 99% compared to 60% for aerated and non-aerated lagoon, respectively. Other emissions were not monitored. The fate of nitrogen in ammonia was not discussed.

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[19] Biochemical Oxygen Demand is used as a surrogate, or indicator, for the amount of biodegradable material in a liquid (or water sample). It is the amount of dissolved oxygen needed by aerobic microorganisms to break down biodegradeable material present in a given water sample at certain temperature over a specific time. The 5-day BOD (BOD_5) measures the amount of oxygen consumed by biochemical oxidation in a sample incubated at 20 °C for a 5-day period. Units are usually expressed as mass of oxygen consumed per volume of sample.

Parker (2008) evaluated the change in odor and volatile organic compound emissions after adding solids separation, covering the primary lagoon, and adding surface aeration to the secondary lagoon at a 3,000 cow flush dairy in Texas. Ambient downwind odor decreased by 80% and volatile organic compound emissions were reduced by 94%. The effect of the aeration by itself was not determined.

Hermanson et al. (1980) measured inlet and outlet nutrients and solids after adding surface aerators to a flushed manure lagoon at Washington State University. Measurements prior to adding the aerator were not done but reduced odor was observed during lagoon storage. However, when the lagoon was vigorously stirred for pumping during cleanout, malodor returned indicating the lagoon had regions of anaerobic activity occurring below the surface aeration zone(i.e., it was a partially aerated system).

Ndegwa et al. (2007) evaluated partial aeration (i.e., provide less oxygen than required for full stabilization) of flushed dairy manure in laboratory reactors simulating aerobic "pretreatment" of manure in order to reduce organic load to, or required volume of, the follow-on anaerobic storage lagoon. After seven days aeration, volatile solids were reduced by 70%. Neither mass balance of oxygen, nor gas emissions were monitored.

In another partial aeration lab scale experiment, Alitalo et al. (2013) treated dairy manure slurry (2.9% TS after mechanical separation) in a set of six tank reactors in which manure flowed from reactor number 1 through number 6 in series with air injected into the bottom of each. The purpose was to reduce carbon and nitrogen in the final effluent. Total carbon content in the final effluent was approximately 50% less than the influent. While gas emissions were not monitored, the authors hypothesized that carbon left the system as carbon dioxide (not methane) because of the aerobic conditions in the reactors.

Sequencing batch reactors (SBR), long used in the waste water treatment industry, may be applicable to dairy manure treatment because of their simple operation scheme and ability to fit into manure collection systems (Lorimor et al., 2006). SBRs operate with intermittent aeration periods (between batch fill and decant stages) which allow reactor conditions to sequence through aerobic, anoxic, anaerobic, anoxic, and aerobic stages during which BOD_5 is reduced (creating carbon dioxide) and nitrification-denitrification[20] occurs (converting ammonia to molecular nitrogen). Cycle times (essentially the same as hydraulic retention time (HRT) can range from about four hours to three days (USEPA, 1999). Wu (2017) used a laboratory SBR to treat flushed dairy manure. Using an 8-hour cycle, VS and ammonia nitrogen were reduced by 96% and 100%, respectively. Gas emissions were not monitored but these authors and Molodovskaya et al. (2008), point out that nitrous oxide emissions are possible if the nitrification-denitrification process is not managed and should not be ignored.

Swine Manure

There is a large amount of literature for aerated swine manure storage lagoons. The majority are related to aeration for odor reduction, but there are some that address GHG reduction, Dennehy et al. (2017) and Maurer et al. (2016). A subset that include GHG reduction measurements or estimates are included here. While the amount of GHG reductions observed

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[20] Nitrification: Biological oxidation of ammonium ion to nitrite (NO_2-), followed by oxidation of nitrite to nitrate (NO_3^-). Denitrification is a microbial process where nitrate is reduced to ultimately molecular nitrogen (N_2).

in aeration of swine manure storage studies (below) cannot be directly applied to dairy manure storage (i.e., as equivalent reductions) due to differences in manure[21] and storage lagoon characteristics, GHG reductions for aerated dairy lagoons are expected (where change in overall nitrous oxide emissions, when accounting for land application of aerated manure, may be small).

Béline et al. (2008) evaluated aerobic treatment of swine waste water at four farms in Brittany (France). All treatment systems used a single above ground open tank that alternated between anaerobic and anoxic phases with HRTs of 30-45 days. Their measurements indicated that, compared to "traditional" management (anaerobic storage for 4-6 months followed by land application) treatment systems reduced ammonia emissions by 30%-52% and GHG emissions by approximately 55%, despite causing an increase in nitrous oxide emissions (1% of N entering system).

Vanotti et al. (2008), using published emission factors and measurements of manure influent and effluent properties, modeled the GHG mitigation potential of an advanced swine wastewater treatment sequence consisting of separation, aerobic treatment, chemical flocculation/precipitation, composting (of solid fraction) and disinfection (of liquid fraction) compared to storage and land application of manure. They estimated that the advanced treatment sequence resulted in a 97% reduction in GHG emissions, attributed to a >99% reduction in methane emissions (compared to anaerobic storage in the baseline scenario) and a 75% reduction in nitrous oxide emissions (attributed to large decrease in nitrous oxide emissions from land application of treated manure).Costs were estimated to be $5.61 per finished pig (in 2006 dollars).

Aerobic Stabilization by Complete Aeration

For complete aerobic stabilization of a storage lagoon, sufficient oxygen delivery and complete mixing is required. Oxygen injection and, especially, complete mixing require energy. Mixing is important to ensure uniformity of temperature and composition throughout the volume, e.g., continuous bulk turnover is needed to eliminate quiescent zones or sludge layers where anaerobic conditions persist. Also, relatively vigorous mixing (high turbulence) prevents clumping of organisms/substrate, and reduces diffusion resistance by thinning the film thickness through which dissolved oxygen must migrate (diffuse) to reach substrate particles and organisms (Cumby, 1987b).

Complete Aeration is Energy Intensive.

Oxygen Delivery

An average dairy cow contributes, through manure excretion, a 5-day biochemical oxygen demand (BOD_5) of 1.3 kg/day (ASABE, 2014). Assuming freestall housing with no solids separation, 80% of the manure is flushed to storage (approximately 1 kg BOD_5 /animal/day). For complete destruction of BOD_5, oxygen must be supplied at a rate of about 1.5 times the BOD_5 load (Zicari et al., 2016). Typical oxygen transfer, or aeration efficiency is 1.5 kg O_2/kWh (Cumby, 1987a), which gives an energy demand for oxygen delivery of 1 kWh/cow/day or 365 kWh/cow/y.

[21] Cows are ruminants and eat forages while swine are monogastrics and eat a diet coarsely similar to humans so manure characteristics are quite different.

Complete Mixing

Specific power input for complete mixing of animal slurries ranges from 10 to 30 W/m^3, which is an average of 76 W per thousand gallons (Cumby,1987b). Dairy lagoon volumes in California, including winter rain and runoff capture, average about 10,000 gallons per lactating cow (Chang et al., 2005; CDFA, 2019). Assuming lagoons on average are 50% full, the volume that needs to be well mixed is 5,000 gallons per cow, which requires 380 W of mixing energy or 3,300 kWh/y, assuming continuous mixing.

Energy Cost - $550/cow/year

Including oxygen delivery, aeration energy is estimated to be 3,665 kWh/cow/y, which would cost $550/cow/y based on average commercial and agriculture electricity price of $0.15/kWh (USEIA, 2019). The 2016 UCD study (Zicari et al., 2016) estimated energy costs of $140/cow/y but seems to have only accounted for oxygen delivery energy (no mixing).

Aerobic Treatment Systems for Lagoon Manure Storage

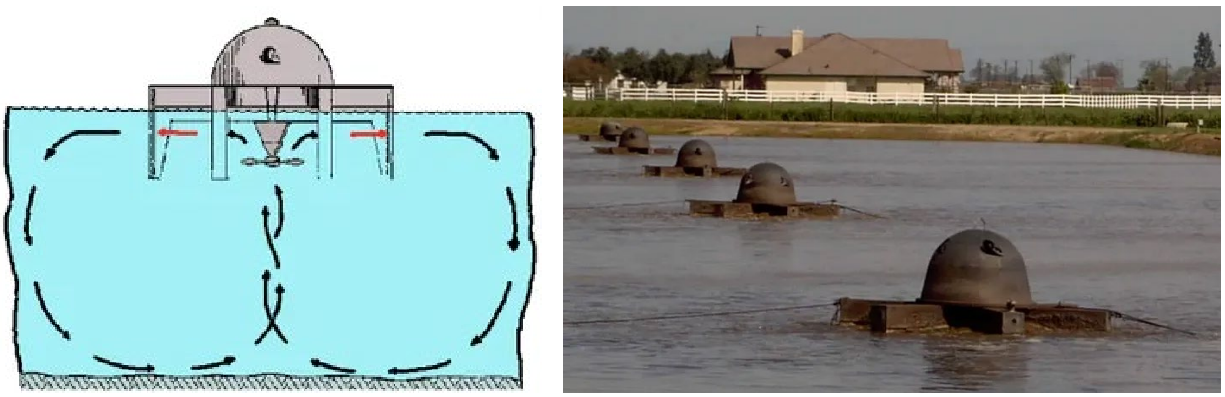
Lagoon aeration systems are, generally, one of two types; 1) surface aeration using floating mechanical mixers and/or air injectors, or, 2) diffusion aeration by sub-surface injection of course and fine bubbles. Subsurface bubble injection without mechanical mixing may not create enough turbulence to keep solids suspended, creating a sludge layer and anaerobic zones at the bottom of the lagoon. In addition, subsurface bubblers require regular maintenance to prevent fouling and clogging from bacteria/biosolids accumulation at the orifices. Because of the low turbulence and fouling issues with sub-surface bubblers, surface mechanical aerators are the most common in manure lagoon settings (Anderson et al., 2014). Three suppliers of surface aeration systems and their costs are discussed briefly below.

PondLift manufactures aerators marketed for odor reduction. They rely on stirring or mixing a lagoon by drawing water up toward the surface where it would absorb oxygen from the air. The units employ a submerged motor and impeller suspended near the surface by a buoy (Figure 4 1). The manufacturer indicates one unit is needed for every 100 animal units (at 1,000 lbs. live weight per animal unit), or about 71 lactating cows[22], cost $11,000 and consume 1.1 kW[23] (PondLift, 2019)).

Figure 4-1 PondLift circulation device in a lagoon

[22] Assumes 1,400 lbs. live animal weight for lactating cow
[23] 1.5 hp electric motor.

Natural Aeration Inc. also manufactures floating lagoon mixers that pull fluid from below upwards to the surface (Figure 4-2). One unit per approximately 90 lactating cows is typical and requires lower VS (or BOD) loading than a typical lagoon (i.e., a larger lagoon is needed) (CalCAN, 2017). Installed cost is around $55,000 each which includes cost to clean out and significantly enlarge the existing lagoon and consumes 1.1 kW each.



Source: Natural Aeration Inc. https://circul8.com/

Figure 4-2 Surface mixer by Natural Aeration Inc.

Aeration Industries International provides aerators to municipal and industrial wastewater treatment and aquaculture markets. The floating aerators provide mixing with direct air injection using a large electric motor to run a high speed mixing propeller and a smaller motor that injects air through the submerged propeller shaft (Figure 4-3). These are designed for sufficient mixing and oxygenation to obtain complete reduction of the wastewater BOD. One unit could treat manure from 200 cows, costs $80,000 (approximately $96,000 installed) and consumes 35 kW electricity (AII, 2019).

Source https://www.aireo2.com/en/

Figure 4-3 Aeration Industries surface-mounted aerator- mixer

System Costs

Costs for the systems discussed above were gleaned from published reports, manufacturer websites, and a manufacturer quoted cost. Per-cow costs (installed, amortized installed, electricity, other operation and maintenance, and total) are shown in Table 4-1. The systems with "mixing only", PondLift and Natural Aeration Inc., have estimated costs of $62 and $147 per cow per year, respectively. The "air injection + mixing" system by Aeration Industries Intl. costs nearly $350/cow/year including $229/cow/year electricity costs due to large mixing and air injection motors.

Table 4-1 Aeration system costs

Source	**Aerator Type**	**Installed Cost ($/cow)**	**Annual Cost ($/cow/year)**			
			Amortized Installed*	**Electricity****	**O&M**	**Total**
PondLift	Mixing Only	$185	$28	$20	$14	**$62**
Natural Aeration Inc. "Circul8"	Mixing Only	$625	$93	$17	$38	**$147**
Aeration Industries Intl. Quote	Air Injection + Mixing	$480	$72	$229	$48	**$349**
Estimate from Zicari et al. (2016)	unknown	$193	$29	$100	$21	**$150**

Notes: * Installed cost amortized over 10 years using interest rate of 8% per year.
**Assumes $0.148/kWh electricity cost.

Aerated Lagoons in California

There are at least two dairies in California that operate an aerated lagoon located in Sonoma and Merced counties. However, the operators were not available to provide information on their system setup and performance.

Field Research Study

To verify actual emissions (emissions reduction) for aerated manure storage on a California dairy, a field study and measurement program would be needed. A suitable and willing dairy would need to allow the installation of an aerator system and visits by research personnel. Ideally, a large anaerobic lagoon would be partitioned in the middle after sludge removal, along a line parallel to the prevailing surface wind direction. The manure inflow system would be modified to feed half the flow to each side of the lagoon partition. After manure has reached a sufficient depth on both sides of the partition, the aerator system would be installed and activated. Emissions measurements for each lagoon portion would then begin. Alternatively, an aerated system could be measured alone (not side by side with an anaerobic lagoon) and emissions compared to those expected without aeration using IPCC (IPCC, 2006);, CARB's Quantification Methodology or other emission factors or modeling. At least two measurement campaigns are recommended; one during the summer and one during the winter.

There are at least two ground-based methods[24] recently used for measuring area source methane emissions: 1) Eddy covariance method (e.g., Runkle et al., 2019), and 2) Open-path

[24] Arndt 2018 also used air borne measurements to estimate methane flux but these were limited to short periods due to budget and logistics, compared to 24/7 availability of the ground-based measurements.

measurement with inverse dispersion modeling (e.g., Arndt et al., 2018; Leytem et al., 2017). An inquiry about the reference studies and costs were pending at the time of publication.[25]

Conclusions

Aeration techniques common at municipal wastewater treatment facilities have been used in manure management, primarily swine manure for odor control, for many years. Lab based studies demonstrate that BOD_5 or VS are reduced by aeration of dairy and pig manure (by more than 96% in one study using a SBR), which implies lower potential methane emissions. Aeration techniques at swine facilities were measured (or modeled) to reduce overall GHG by approximately 55% in simple open aerated tanks, to more than 99% for a sophisticated aerobic treatment system that replaced a storage pond. Though more nitrous oxide is emitted by aerated treatment devices than from anaerobic lagoons, overall system nitrous oxide emissions are likely lower because there is less nitrogen in the treated material used in land application (lower nitrous oxide emissions from land application at least for treated swine manure). Manure is used for crop fertility. If molecular nitrogen is the end product of aeration, then there is reduced organic and ammonia nitrogen available for land application with a net loss of nutrients. This creates an inefficiency of its own if solids are then applied with lower total nitrogen available for crops. On some dairies, there is too much nitrogen, so locally nitrogen loss may be an advantage, but it is not clear if this is a net benefit regionally.

Complete aeration (oxidation) of an anaerobic lagoon is energy intensive primarily because of the complete-mix requirement and large volume of a storage lagoon. It is unclear if surface aerators, used in odor control, can provide sufficient mixing.

A research study to measure emissions from an aerated lagoon is achievable using ground-based area source measurement techniques. A preliminary cost estimate for this research is about $450,000, not including the cost of an aerator system and any needed lagoon modification.

Recommendations:

There is not enough information regarding the performance of aeration systems in dairy lagoons to warrant adding to the AMMP Quantification Methodology at this time. A research study to measure and model emissions from aerated lagoons is recommended. This should include monitoring nitrogen fate and benefits or impacts due to changes in nitrogen in the land-applied treated manure.

[25] A preliminary cost estimate for a field research program to monitor lagoon emissions using eddy covariance is approximately $500.000 for a 1-2 year study period (includes equipment cost of $300,000).(Suvočarev, 2019).

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5. Evaluation of Livestock Manure Pyrolysis and/or Gasification Systems

Conversion Pathways

Conversion of organic material can proceed along three main pathways—biochemical, thermochemical, and physicochemical.

Biochemical conversion processes include anaerobic digestion (AD) and/or fermentation (where methane and carbon dioxide are primary products of AD. Liquids such as ethanol and other biochemicals as well as carbon dioxide also are products of fermentation. Aerobic conversion, such as composting, produces a more or less stabilized organic material and carbon dioxide.

Biochemical conversion proceeds at lower temperatures and lower reaction rates, which can require large volume reactors (digesters or compost facilities) if large amounts of material are processed. Higher moisture feedstocks are generally good candidates for biochemical processes.

Physicochemical conversion involves the physical and chemical synthesis of products from feedstocks and is primarily associated with the fatty acid methyl ester (FAME) biodiesel production process. Renewable diesel, derived from waste lipid materials combined with externally derived hydrogen, is also included in this category.

Thermochemical conversion processes include combustion, gasification and pyrolysis:

Combustion is the complete oxidation of the fuel for the production of heat at elevated temperatures without generating commercially useful intermediate fuel gases, liquids, or solids. The process employs excess oxidizer to ensure complete fuel conversion, but also can occur under fuel rich conditions. Products of combustion processes include heat, oxidized species (e.g. carbon dioxide, water), products of incomplete combustion and other reaction products (most as pollutants), and ash. Other processes, such as supercritical water oxidation can produce similar end products at lower temperatures.

Gasification refers to the conversion of a solid or liquid feedstock into an energetic, or fuel, gas (often called producer gas or synthesis gas). Autothermal gasification uses partial oxidation of the substrate, using substoichiometric air or oxygen, to produce principally carbon monoxide, hydrogen, methane, and light hydrocarbon gases in association with carbon dioxide$_2$, molecular nitrogen, and water vapor depending on process used. Allothermal, or indirect, gasification uses an external heat source for the gasification reactor. Gasification processes also produce liquids (tars, oils, and other condensates) and solids (char, ash). Fuel gases can be used in internal and external combustion engines, fuel cells, and other prime movers, or as a chemical feedstock for other products including liquid fuels.

Pyrolysis is a process similar to gasification except generally optimized for the production of fuel liquids (pyrolysis oils) or char solids (biochar). Pyrolysis also produces gases. Pyrolysis thermally degrades (or decomposes) material without the addition of any air or oxygen.

Thermochemical conversion is characterized by higher temperature and conversion rates. It is best suited for lower moisture feedstocks and is generally less selective for products.

Thermal Conversion of Manure

Fresh, scraped, or flushed manure has too much moisture for most thermal conversion processes.

"Net thermal energy" or the energy remaining after evaporating moisture from dairy manure is shown in Figure 5-1 as a function of TS.[26] For flushed manure, where (TS are typically 1-3%, there is not enough energy to evaporate the 97-99% moisture (net energy is < 0). Scraped manure, at around 11-13% TS has a net energy of about 0.

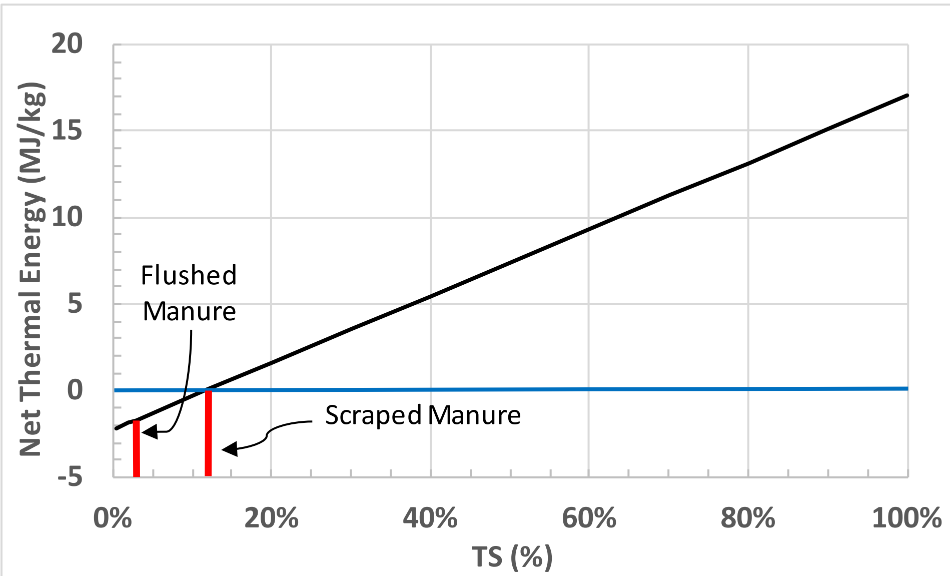

Figure 5-1 Dairy manure net thermal energy vs. total solids

For thermal conversion of dairy manure (other than hydrothermal or "wet" gasification or oxidation), solar evaporation or solid / liquid separation with follow-on solids drying would be required to obtain a feedstock suitable for most thermal conversion systems.

If solids separation or evaporation is used to prepare the feedstock for gasification or conversion, rather than flushing to lagoon storage, then this activity would be responsible for any GHG reduction due to diversion from lagoon. The gasifier (or thermal conversion) would only be responsible for any additional GHG reduction due to its products displacing fossil or non-renewable products, such as natural gas combustion or grid electricity (an estimate is calculated below).

Review of the Literature

A search for entities engaged in thermal conversion of manure was conducted including grey and peer reviewed literature, and websites. While there is abundant information related to

[26]Energy content of dairy manure, is approximately 17 MJ/kg (higher heating value, dry basis) and is ~20 MJ/kg dry ash-free basis (Carlin et al., 2007 & ASABE, 2015).

academic and laboratory studies of manure conversion technologies, properties of feedstocks and outputs (i.e., biochar and other solids), only two companies were found that are currently active in thermal conversion of cattle or dairy manure; Coaltec and Agricultural Waste Solutions. In addition, several poultry litter combustion systems are operating (or were recently) in the Chesapeake Bay Watershed as part of an effort to improve phosphorous management in the watershed. Brief descriptions of these two technologies and the Chesapeake Bay demonstrations follow. In addition, hydrothermal conversion, an interesting emerging technology for high moisture feedstocks is discussed. Finally, a discussion on biochar as a source of nutrients and ability to sequester carbon are provided.

Coaltec

Coaltec markets gasifier-close-coupled-combustion systems (sometimes called staged combustion where producer gas is immediately burned without intermediate processing or cleaning) for manure drying and disposal, heat generation, and possibly biochar co-product. Four systems using manure as the primary feedstock have been installed in the US (none in California); two at dairies (Ohio and Indiana) and two at poultry farms (McGolden, 2019). The dairies with Coaltec systems use scrape and/or vacuum manure recovery from the feedlanes. Recovered manure TS is approximately 15-20% which includes some bedding material or spilled feed. Moisture content for gasifier feedstock should be less than 40% (wet basis) and ideally 15% (or TS = 85%) (Kowalczuk, 2012).

The Coaltec system uses heat released from burning the producer gas to dry the incoming manure so that it can be gasified (Figure 5-2). After drying, the manure is fed to the gasifier along with sub-stoichiometric air where partial oxidation occurs to supply heat to drive the gasification reactions. The combustible producer gas flows directly to a combustor or thermal oxidizer (with no intermediate gas processing or cleaning) where it is burned. Heat from the combustor is used to dry the manure. Excess heat can be used to dry more manure for use as bedding and/or provide heat for animal housing heat in cold climates. This produces criteria and toxic air contaminants. Uncontrolled NOx emissions (as fuel NOx) can be high because of relatively high nitrogen content of manure. This would likely make obtaining an air permit in California, especially the San Joaquin Valley, challenging.

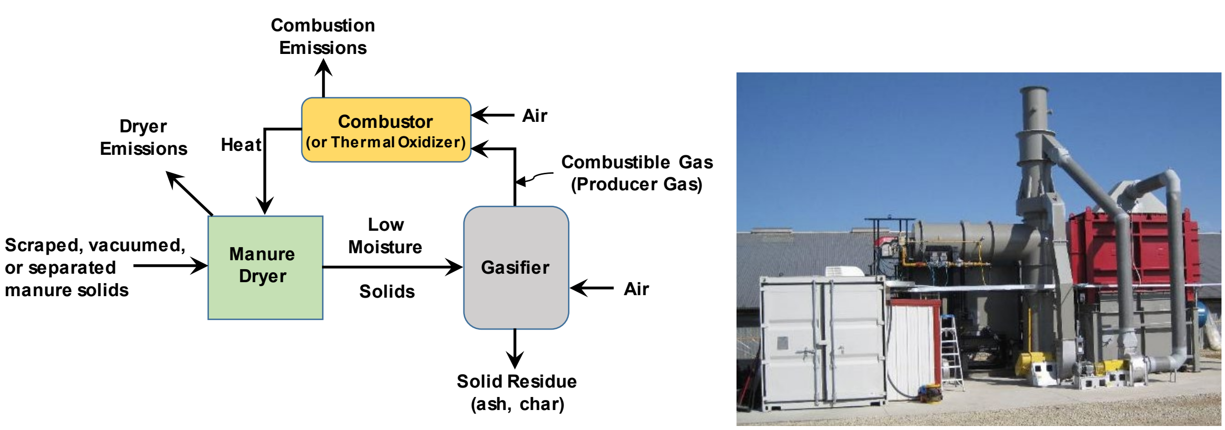

Figure 5-2 Schematic and photo of the Coaltec gasifier-close-coupled-combustion system.

Agricultural Waste Solutions

Agricultural Waste Solutions, Inc. (AWS) has demonstrated production of renewable diesel from dairy manure at the Scott Brothers Dairy in Riverside County.[27] The process follows the so-called biomass-to-liquid (BTL) pathway where low moisture solid biomass (in this case, dairy manure at 35% moisture) is gasified followed by extensive cleanup of the raw gas, gas reforming to adjust the carbon to hydrogen ratio in the synthesis gas, and finally production of liquid and wax hydrocarbons through the Fischer-Tropsch process (Figure 5-3).

The AWS commercial model for a small, 10 barrel per day (153,000 gallons per year) facility using manure from 3,000 dairy cows would produce renewable diesel for $5 per gallon (McCorkle 2016). The financial model also assumes 20 tons per day of biochar is produced and sold for $270/ton. Without biochar sales, the renewable diesel minimum selling price would be more than $17 per gallon. Fuel yield is about 50 gallons per cow per year or 14 gallons per bone dry ton (BDT) of manure. This is fairly low yield compared to modeled and expected yield for BTL processes of 40-55 gallons per BDT in the literature (Anex et al., 2010; de Jong et al.,2015; Zhao et al., 2015).

AWS is looking for funding to continue development and is exploring biochar markets (LA County, 2019).

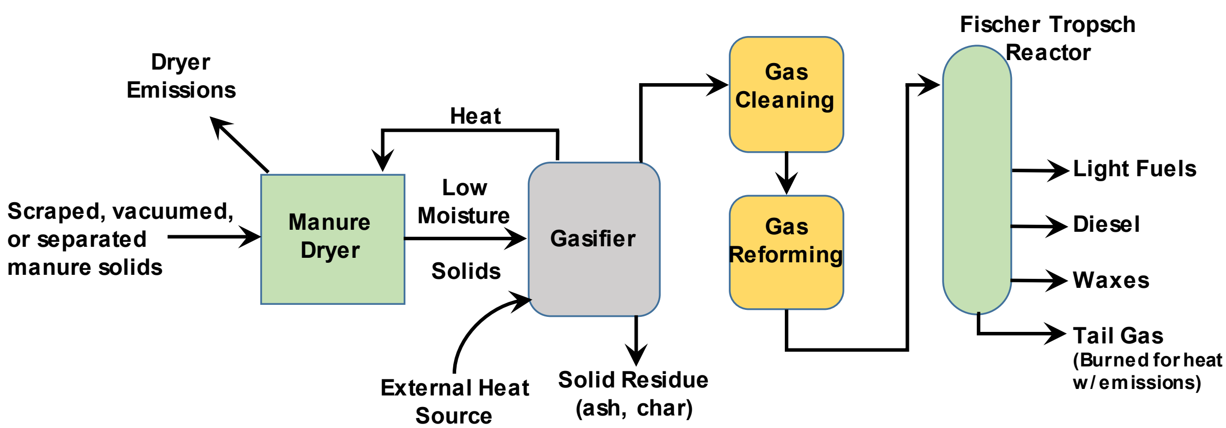

Figure 5-3 Schematic of the AWS manure-to-renewable diesel process.

Poultry Litter Projects in Chesapeake Bay Watershed

Sustainable Chesapeake and other organizations in Chesapeake Bay Watershed evaluated the use of thermal technologies to convert poultry litter to heat or electricity. This was in support of efforts to reduce manure phosphorus load to the watershed.

Several thermal manure-based energy systems were installed and assessed at five farms. They were all simple solid fuel combustion systems, with one being a two stage combustor (gasifier-close-coupled- combustion, see Table 5-1).

As poultry litter has a moisture content 25-35% (UGCE, 2011), each system was able to accept the feedstock with no additional drying. They all provided heat for the brooding houses, supplementing conventional heat sources (usually natural gas or propane).

[27] Funded in part by a $658,000 grant from the California Energy Commission (ARV-10-43).

Ambient air quality in the region was such that no controls were required for criteria pollutant emissions (see emissions in Table 5- 3).

Table 5-1 Chesapeake Bay Watershed Farm to Energy Demonstration Facilities

Technology Name	Description	Feedstock	Energy use	State
Bio Burner 500	Fixed grate solid fuel combustion	Poultry Litter	Heat for brooding house	Virginia
Global Re-Fuel PLF-500	Vibrating grate solid fuel combustion	Poultry Litter	Heat for brooding house	West Virginia Pennsylvania
Blue Flame Boiler	Stoker- moving chain grate solid fuel combustion	Poultry Litter	Heat for brooding house	Pennsylvania
Ecoremedy-gasifier/combustor	Moving grate updraft gasifier-close-coupled- combustion	Poultry Litter	Heat for brooding house	Pennsylvania

Hydrothermal Conversion

Hydrothermal conversion, though technically challenging, may have potential for manure conversion because it's amenable to high moisture feedstock and may offer a pathway for direct GHG reductions without first drying or separating solids. Hydrothermal processes occur in liquid water at temperatures from ~ 150 °C-400°C at or above the saturation pressure (1.5 MPa to > 22 MPa [215 to > 3200 psi]). Depending on the temperature and pressure, char production can be optimized at around 200°C, bio oil production or "liquefaction" (Hydrothermal Liquefaction or HTL) at intermediate temperatures (300-350°C), and hydrothermal gasification (wet gasification) using catalysts over 200-400°C temperature range, or gasification without catalysts above the critical point for water (374°C and 22.06 MPa), or supercritical water gasification (Awasthi et al., 2019).

Operating above the critical point with addition of oxygen to the reactor allows for oxidation of the substrate (so called supercritical water oxidation or SCWO) producing water, carbon dioxide and heat. SCWO has been investigated and used for over 35 years to treat aqueous and hazardous organic wastes with current commercial development ongoing in the municipal waste water treatment industry (Adar et al., 2019).

Hydrothermal Liquefaction of Manure

Nearly all the work in manure HTL is at lab scale (Fan et al., 2018). For example Theegala & Midgett (2012) produced about 5 g of bio-oil from 20 g samples of separated manure solids that had been dried and pulverized and then rehydrated to 20% TS. Yin et al. (2010) used HTL to convert about 50% of the mass of carbon in cattle manure slurry to bio oil (biocrude) with remainder as gas and solid char. Tushar et al. (2016) produced hydrogen and methane via SCWG of bio oil from 10 g quantities of cow manure. Similar research includes Toufiq Reza, et al. (2016), and Dai et al. (2015).

Researchers at Cornell University evaluated the economic feasibility of using a wet gasification technology to reduce mass of manure that needs to be disposed of or spread on nearby fields and reduce the cost of manure management. Working with a technology provider (not named), the economics were modeled for a 2,500 cow dairy in New York. Though specifics are not given, the conclusion was that the technology would not be economic except under the most optimistic (but unlikely) assumptions for energy and solids sales price (Wright, 2019).

Commercial HTL Activity

Genifuel, using hydrothermal liquefaction technology licensed from the Pacific Northwest National Laboratory, is pursuing commercial scale projects at municipal wastewater treatment plants including the Central Contra Costa Sanitation District. The process operates at 350°C and 21 MPa (3,045 psi), just below the critical point, and is expected to produce methane and a biocrude that can be upgraded into liquid fuel. Nominal yield is 450 kg bio oil and 66 kg methane per dry ton of biosolids (Oyler, 2017).

Biochar from Manure

The US biochar industry is comprised of some 135 biochar producers. About 45,000 tons per year of biochar is produced, almost all from woody biomass. Most is sold to agricultural applications including field and orchard crops, horticulture, gardens, and landscaping. Some is used in filtration or odor control (Draper et al., 2018).

Coaltec, described previously, can adjust operating conditions of their gasifier-combustor to leave more unreacted carbon in the solid residue, i.e., more biochar output. Consequently, biochar is listed as a byproduct in their marketing literature. No other manure derived biochar producers were found.

From 2005-2010, Virginia Tech with BioEnergy Planet, attempted to commercialize a mobile fast pyrolysis system to convert poultry litter to bio oil and char. Chevron Oil provided funding (CBC, 2012). Apparently the system was not feasible, either technically or economically as the only published information seems to be from lab based work at Virginia Tech using a 2 inch diameter fast pyrolysis reactor with feed rate of 0.2 kg/h (Kim et al., 2009; Agblevor et al., 2010; Mante & Agblevor, 2010; Mante & Agblevor, 2012).

Biochar – Nitrogen, Phosphorus, and Sequestration

With thermal conversion, most of the nitrogen present in the feedstock is released in gaseous form as hydrogen cyanide, ammonia , nitrogen oxides, and molecular nitrogen while almost all of the phosphorus is recovered in the ash or biochar (Turn et al., 1998).

Regarding availability of phosphorous, the literature review by Brod (2018) indicates that thermal treatment reduces phosphorous availability compared to untreated manure. The reduction in availability is higher with corresponding higher conversion temperature (i.e., phosphorous in char produced at above 600°C was transformed mostly to relatively stable calcium phosphates), though depending on soil pH and effect of weathering (time), phosphorous eventually becomes plant-available. Brod (2018) concludes that biochar produces a slow-release fertilizer (for phosphorous) rather than an effective substitute for untreated manure or mineral fertilizers.

Biochar can be valuable as a soil conditioner enhancing plant growth by retaining or adding nutrients, improve moisture holding ability and other soil properties. In the right circumstances, biochar can be a means of carbon sequestration but in other cases it can increase soil carbon mineralization (carbon dioxide release), nitrous oxide, and methane emissions. Whether biochar enhances soil organic matter and sequesters carbon or accelerates soil carbon mineralization is complex and depends on soil pH, microbial community, initial soil organic matter content, and other factors (Ding et al., 2018; Sheng & Zhu, 2018; De la Rosa et al., 2018; De la Rosa et al., 2018b).

A Gasification System that may be Feasible in California

Required Energy Inputs

As mentioned above, a gasifier does not work well with high moisture feedstocks. In general, 15-30% moisture content in feedstock is optimal with up to 40% acceptable (Williams and Kaffka, 2015). To use flushed or scraped dairy manure as a feedstock, a large amount of moisture would need to be removed. In fact, the energy needed to evaporate the moisture is equal to or more than the energy content in the solids. Therefore, to obtain a useable feedstock for thermal conversion (gasification or combustion), scraped or slurry manure would need to be spread and solar dried on a large pad, or a solid separator is needed to recover feedstock solids from flushed or scraped manure. While moisture content of separated manure solids (from a screen separator) is about 80% (Edalati et al., 2019), there would be enough thermal energy in the exhaust from a gasifier-engine-generator or a combustor to pre-dry fresh separated solids down to about 40% moisture.

Net Energy Production

For recoverable solids from flushed dairy manure for thermal conversion feedstock, assume 80% of manure is flushed (Quantification Methodology default assumption of 20% deposited on land) and 35% solids separation efficiency for sloped screen separator (see Chapter 2 recommendations). This yields 38% of excreted manure solids as recoverable for gasifier feedstock (which is about 2.6 kg TS per lactating cow per day)[28]. The energy content of dairy manure is about 16.9 MJ/kg (HHV, dry basis)[29], so the energy in the recovered solids is 43.5 MJ/cow/day. The net thermal efficiency of electricity production from small gasifier-engine-generator systems (Figure 5-4) is about 20% (Williams and Kaffka, 2015), so 8.7 MJ or 2.4 kWh of electricity could be produced per cow per day (0.1 kW per cow).

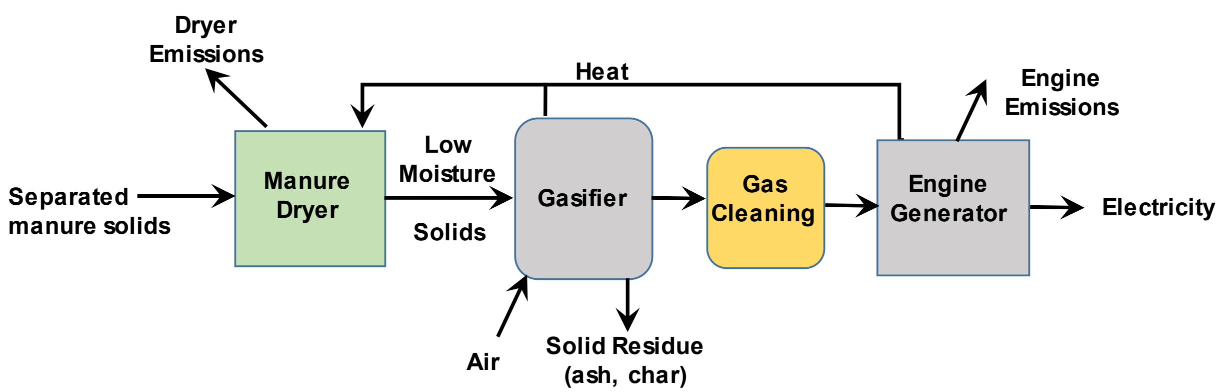

Figure 5-4 Schematic of a manure gasifier-to-engine generator system.

[28] From the CARB Quantification Methodology; a lactating cow excretes 7.76 kg VS per day. For an average VS/TS of 0.843, TS/cow/day = 9.2 kg.
[29] Dairy manure VS has a heating content of ~20 MJ/kg (dry, ash-free basis). The energy of TS, therefore, is VS/TS * 20.

Costs

Table 5-2 displays capacity, capital cost, and levelized cost of energy (LCOE)[30] for a manure gasification system over a dairy size range of 500 to 10,000 cows. The Black & Veatch small gasifier LCOE calculator (B&V, 2013) developed for the California Public Utilities Commission (CPUC) was adapted for a range of capacities (50-1,000 kW) and assuming a zero feedstock cost.

The estimated LCOE (176 to 208 $/MWh) brackets the current BioMAT tariff (electricity price) for dairy projects of $188/MWh (CPUC 2019). LCOE would be greater for non-zero feedstock costs (i.e., if solids separation and drying costs were included in the project).

Table 5-2 Capacity, cost, and LCOE for dairy solids gasification-to-electricity

Dairy Size (cows)	Capacity (kW)	Capital Cost		LCOE ($/MWh)
		($/kW)	Total Installed Cost ($)	
500	50	9,036	451,790	208
1,000	100	8,431	843,069	200
5,000	500	7,177	3,588,694	183
10,000	1,000	6,697	6,696,739	176

Notes: The LCOE calculator developed by Black & Veatch for the CPUC, based on a 3 MW electric capacity gasifier-engine-generator with emission controls, was used. It was adapted by scaling capital cost with size ($6000/kw for 3,000 kW size in the B&V model, scale factor 0.9) and setting feedstock cost to zero (though feedstock cost would be non-zero if solids separation and drying equipment were installed as part of the project).

Criteria Pollutant and Hazardous Emissions

While emissions of the gas producing unit, or reactor, for most autothermal gasifier systems can indeed be zero[31], the system will produce typical combustion emissions if the product gas is burned onsite for heat or in a prime mover (engine or gas turbine) for electricity. Because of the relatively large amount of nitrogen in dairy manure and poultry litter (2-5% on a dry weight basis) (Chastain et al., 2001 & 2003), NOx emissions would be significantly greater than for gasifiers (or combustors) fueled by woody biomass which typically has less than 0.5% nitrogen (Francescato et al., 2008).

Nitrogen in solid fuel (fuel-N) reacts during gasification to form, among other products, ammonia and hydrogen cyanide, which are NOx precursors. In gasification processes, the fraction of fuel-N transformed to ammonia and hydrogen cyanide in the product gas is 20-30%, with about 60% of fuel-N leaving as molecular nitrogen (N_2) and 10-15% is found in the char and tar (Aznar et al., 2009: Yu et al., 2007).

Uncontrolled criteria pollutant and hazardous air pollutant (HAP) emissions from the poultry litter gasifiers and combustors demonstrated in the Chesapeake Bay Watershed Farm to Energy program are displayed in Table 5-3.

[30] LCOE is the energy sales price the project needs in order to be economically viable (i.e., required revenue).

[31] i.e., air or oxygen fed partial oxidation reactor operating under slight vacuum will have no emissions.

Table 5-3 Emissions from poultry litter combustion and gasification-close-coupled combustion systems (lb/MMBtu)

NOx	0.1 - 0.72
SO2	0.17 - 0.7
CO	0.14 - 0.55
PM	1.38 - 2.99
HCL	0.06 - 0.39
VOC	8.2E-07 - 8.1E-03
HAPs	0.05 – 0.72

For comparison, uncontrolled NOx emissions from wood-fired boilers and low-NOx natural gas combustors are 0.22-0.49 and 0.02-0.08 lb/MMBtu, respectively. Hazardous air pollutant emissions from wood-fired boilers and natural gas combustion are 0.04 and 0.002 lb/MMBtu, respectively (USEPA, 2003).

With funding from a small business innovation research grant (SBIR), Community Power Corporation demonstrated electricity production from an engine-generator burning producer gas from a small downdraft gasifier (autothermal) fueled by poultry litter. Noting the system was not optimized for poultry litter, uncontrolled engine NOx emissions were measured at around 5,000 ppm which equated to 180 lbs-NOx/MWh, or about 8 lb-NOx/MMBtu of solid poultry litter input (Reardon et al., 2001). Follow-on funding produced a gasifier design specifically for poultry litter which was operated for a short period, flaring the gas. No emission measurements were reported (Reardon & Lilley, 2004). The project apparently did not progress further.

While there are no known manure gasifier and engine-generator systems installed in California, a number of woody biomass gasifier and engine-generator systems have been installed and permitted. A recent permitting example is for the Cabin Creek project near Truckee which obtained an authority to construct from Placer County Air Pollution Control District in 2015 for a 1 MW "syngas fired" engine with SCR/Oxidation Catalyst for emissions control (Placer County ATC: AC-14-30A). Because of the higher nitrogen content in manure compared to woody biomass (as mentioned above), nitrogen compounds (especially ammonia) in the product gas would best be removed, or reduced, before induction to the engine, adding costs. It is unclear whether a manure gasifier with engine generator could meet NOx emission limits in the San Joaquin Valley Air Pollution Control District.

Greenhouse Gas Emissions Reduction

Because the gasifier system requires separated, and partially dried solids, the only GHG benefit that can realistically be ascribed to the gasifier is that due to displacing average grid electricity with renewable (assumed carbon neutral) electricity produced by the generator set. The solid separator (or pad-dried scraped manure) is responsible for diverting solids from lagoon storage with concomitant GHG reduction.

Diverting 35% of flushed solids from lagoon storage results in GHG reduction of 2.05 $MTCO_2e$/cow/year (AMMP Quantification Methodology using 35% screen efficiency). Annual electricity production per cow is 0.88 MWh which displaces 0.2 MT CO_2e/ year of grid electricity emissions (where average grid emission factor is 0.2279 MTCO2e/MWH; CARB, 2017). Total GHG reduction is 2.25 MT CO_2e/cow/year with about 9% due to electricity production (Table 5-4).

Table 5-4 GHG reduction due to solids separator and onsite gasifier-to-electricity production

Dairy Size (cows)	GHG Reduction ($MTCO_2e/y$)		
	Solids Separator	Renewable Electricity Production	Total
500	1,025	100	1,125
1,000	2,049	201	2,250
5,000	10,246	1,005	11,251
10,000	20,493	2,010	22,503

Conclusion

Evaporation or solid/liquid separation with follow-on solids drying is required to obtain a suitable feedstock for most thermal conversion systems. The activity/device responsible for diverting volatile solids from lagoon storage in a gasifier "project" is the solids separation and/or evaporation.

The gasifier (or thermal conversion) would be responsible for any additional GHG reduction due to its products displacing fossil or non-renewable products, such as natural gas combustion or grid electricity.

Recommendation

We do not recommend including gasification or thermal conversion in the AMMP Quantification Methodology.

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Appendix 1

A literature search using "Web of Science"   [Web of Science [v.5.29] - Web of Science Core Collection Basic Search] literature database was conducted for time budgets of dairy animals. The search term used was ("Time budget*" AND dairy) which resulted in 122 hits. Based on titles and abstracts, it was determined that 109 of the articles were not relevant and 11 were reviewed and used in this chapter.

Table A-1. Breakdown of article topic from ("Time budget*" AND dairy) search term results

Was not about dairy cows	17
Dairy design and/or environment effects	15
Lameness	14
Social behavior	14
Resting and Lying time budgets	14
Care and comfort	12
Effects of feed on time budget	10
Pasture dairies	9
Milk yield	4
Total Not Relevant	**109**
Reviewed	13