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Review

Biogas recovery by anaerobic digestion of Australian agro-industry waste: A review

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ABSTRACT

On-farm, intensive feeding and processing in Australian red meat, dairy and pork industries produce substantial quantities of organic waste (~79 GL·annum^{-1} liquid waste plus ~2 megatonnes·annum^{-1} solid waste) and waste management is a major cost (~180 million Australian dollars. annum^{-1} for red meat plus dairy processing). Anaerobic digestion can instead extract value from organic waste as biogas energy and biofertiliser to reduce operational costs and environmental impacts, and to improve industry profitability. Understanding key information gaps is a fundamental step towards fully realizing profitable opportunities for anaerobic digestion. This is addressed here via a critical evaluation of available information on Australian agro-industries (specifically dairy, pork and red meat), their waste availability, biogas energy potential, and potential anaerobic digestion approaches. The analysis revealed varying extents of information, but good biogas energy potential (~13.8 PJ·annum^{-1}) to meet a significant energy demand (~18 PJ·annum^{-1}). Waste management within respective agro-industries influenced waste amounts and characteristics, which affected anaerobic digestion options. Anaerobic co-digestion, involving aggregated digestion of two or more waste types within or across industries, can provide further opportunities by boosting biogas production and harnessing spare digestion capacity. Overall, cross-industry collaboration, policy support and technology development could help harness the significant opportunities for aggregated biogas production in Australian agro-industries.

Contents

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1. Introduction

Australia's energy consumption for the year 2017/18 was approximately 6,172 PJ (PJ) (Department of Environment and Energy, 2019). Whilst this energy mostly originated from oil (39%), coal (30%) and natural gas (25%), renewable energy will play an increasingly important role in Australia's energy future, with a steady growth in renewables over the past decade (Department of Environment and Energy, 2019). The Australian agricultural sector is a comparatively minor energy consumer (117 PJ; 1.9% of total, year 2017/2018 (Department of Environment and Energy, 2019)) but a significant emitter of greenhouse gasses (GHG), i.e. up to 14.1% of Australia's GHG emissions (Commonwealth of Australia, 2018). Enteric fermentation dominates these emissions, but emissions from manure management are also important at 9.7% of total agricultural emissions in 2018 (Commonwealth of Australia, 2018). This is an opportunity because methane (in biogas) from manure management can instead be captured and used as a renewable energy source, thereby significantly reducing carbon footprint of agricultural production. For example, such carbon reduction opportunities have been demonstrated for the Australian pork sector by Wiedemann et al. (2016).

Anaerobic digestion (AD) is a natural biological process that occurs in the absence of oxygen and converts organic matter into biogas. Biogas is a mixture of mostly methane and carbon dioxide (CO_2) with some trace gases (Mata-Alvarez et al., 2014). The methane in biogas can be used to produce renewable heat, electricity or cooling (Sibilio et al., 2017), thereby reducing dependence on fossil fuel energy. Anaerobic processing of agro-industrial waste can also significantly value-add because waste management is a major cost to agro-industries. For example, waste management costs in the red meat sector have been estimated at ~100–200 M Australian dollars (AUD) per annum (O'Hara et al., 2016) and for milk processing in Victoria (VIC) representing the majority of milk production in Australia, waste management costs have been estimated at ~37 M AUD in 2003 (Allinson et al., 2007). To date, production costs (specifically energy and waste processing) have predominantly driven the adoption of AD in Australian agro-industries, with regulatory requirements (e.g. odour mitigation) also being very important but a secondary driver. Anaerobic processing of organic waste into biogas energy can be used to offset operating costs and thereby improve the profitability of agro-industries. Revenue streams can also be derived from gate fees for receiving and processing organic waste that would otherwise have been disposed to landfill (Edwards et al., 2015).

Australian agro-industries produce a diverse range of organic waste streams relevant to biogas production, and these waste streams vary significantly between and within respective agro-industries. This is partly due to distinct onsite production, processing and waste management practices, which differs and therefore produces different types of waste, such as in dairy on-farm (Birchall et al., 2008), beef feedlots (Meat and Livestock Australia, 2012), piggeries on-farm (Tucker, 2018), milk processing (Liu and Haynes, 2011) and meat processing (Jensen et al., 2014). Tables 1 and 2 summarise important waste characteristics and show that agro-industry organic waste can be broadly classed as liquid or solid waste. This is important, because it influences feasible AD technologies and approaches (Li et al., 2011) (See Section 3). Various organic waste streams also exhibit distinct biochemical methane potentials (B_0) depending on their macro-composition (i.e. carbohydrate, protein and fat content) (Angelidaki and Sanders, 2004). B_0 can also be influenced by ageing of waste when stored for extended periods onsite, as has been previously observed for beef feedlot manure (Gopalan et al., 2013b) and with pig and dairy manure (Moller et al., 2002). In cases, seasonal or batch-wise production of certain organic waste streams may influence the need for onsite storage and suitability of batch vs. continuous AD approaches (See e.g. pork, Section 3.4). Batch vs. continuous AD process selection is also influenced by waste characteristics. For example, due to kinetic considerations, batch AD can be volumetrically more efficient than continuous AD in cases where the overall digestion rate is limited by hydrolysis (Batstone and Jensen, 2011). Continuous solid phase AD is generally more expensive than batch solid phase AD, because of the need to add and remove waste during operation of a continuous digester (Batstone and Jensen, 2011).

Anaerobic co-digestion (AcoD) is the simultaneous AD of two or more feedstocks and is a common strategy to overcome the limitations experienced with mono-digestion of single substrates, by balancing feedstock composition and moisture, diluting inhibitors, or by regulating digestion pH (Mata-Alvarez et al., 2014). For example, this would be important where influential chemical inhibitors are found in agro-industrial waste, e.g. sodium, ammonia and fat, oil & grease (FOG) (See Section 3). Moreover, the availability of essential micro and macro-nutrients will also be important to ensure that AD and/or AcoD is stable and achieves optimal performance (Romero-Guiza et al., 2016).

AcoD can mix carbon-rich waste substrates with nitrogen-rich substrates. The carbon-rich substrates can provide easily biodegradable organic matter (e.g. glycerol (Astals et al., 2013)) to boost organic loading and to provide labile carbon that induces a so-called "priming effect" of the digestion microbiology as described by Insam et al. (2016). Nitrogen-rich substrates can assist with pH buffering to prevent acidification when highly biodegradable co-substrates ferment during AD and can also facilitate a macro-nutrient balance to stabilise AD performance (Hagos et al., 2017). AcoD also increases organic loading (more organic matter can produce more biogas energy, up to a certain organic loading limit) to better utilise spare capacity at existing AD facilities (Nghiem et al., 2017). This is important to achieve scale of energy production, to improve economic feasibility, and to facilitate cross-industry collaborations by importing waste from one sector to another sector for AcoD (with due consideration of biosecurity) (See Section 5).

The biogas energy potential in Australian agro-industry organic waste is substantial (see Section 4). However, anaerobic treatment of agro-industrial organic waste in Australia is still in the early stages of adoption (Edwards et al., 2015). For example, in 2019, the total number of AD plants was estimated at around 242 and the majority of these were municipal sewage sludge digesters and landfill gas facilities (IEA Bioenergy Task 37, 2020). Agricultural AD plants were the minority, mainly using manure from piggeries (20

Table 1
Typical composition of various agro-industrial liquid waste types, given as a range of reported values or as individual values where only single values were found.

Characteristic[a]	Dairy farm effluent[b]	Dairy processing[c]		Piggery effluent[d]	Red meat processing combined effluent[f]
		Combined processing effluent	Whey		
Total COD ($mg \cdot L^{-1}$)	438-5,044	50-95,000	35,000–128,300	22,000–96,000	840-24,200
TS ($mg \cdot L^{-1}$)	800–27,000	101-5,100	3,190–73,200	15,000–69,000	500-8,396
VS:TS ratio	0.8–0.86	–	0.84–0.93	0.63–0.89	0.63–0.78
TSS ($mg \cdot L^{-1}$)	221-2,996	10-12,500	1,300–22,150	–	1,000–6,830
VFAs ($mg \cdot L^{-1}$)	–	–	–	200-7,500	130–770
FOG ($mg \cdot L^{-1}$)	–	2-4,890	350-1,100	–	5-4,570
TN or TKN ($mg \cdot L^{-1}$)	100–506	5–830	10-1,460	800-4,200	39–1000
Total K ($mg \cdot L^{-1}$)	164–705	15–60	1,430–30,500	124–784[&]	20–150
Total P ($mg \cdot L^{-1}$)	17–82	0.02–160	124-8,300	70-1,700	20–108
Total S ($mg \cdot L^{-1}$)	17–65	–	1,000	9[&] – 143.77	–
pH	7.1–8.22	4.0–12.5	3.8–7.12	7.0–8.7[&]	6–8.4
B_0	180–250 $m^3_N\ CH_4 \cdot t^{-1}$ VS_{added}	72–90% COD reduction	264–424 $m^3_N\ CH_4 \cdot t^{-1}$ VS_{added}; 16.5–22.7 $L_N\ CH_4 \cdot L^{-1}$ whey	150-640[e] $m^3_N\ CH_4 \cdot t^{-1}$ VS_{added}	2.6–4.8 $m^3_N\ CH_4 \cdot m^{-3}$ effluent [g]

The range of values given represent data reported by
[a] COD, Chemical oxygen demand; FOG, fat, oil & grease; TS, Total solids; TSS, Total suspended solids; TKN, Total Kjeldahl nitrogen; TN, Total nitrogen; TN, Total nitrogen; Total K, Total potassium; Total P, Total phosphorus; Total S, Total sulphur; VS, Volatile solids; VFAs, Volatile fatty acids; CH_4, methane.
[b] (Birchall et al., 2008; Craggs et al., 2008; Fyfe et al., 2016; Jacobs et al., 2008; Longhurst et al., 2000; Mason, 1997; Phelps et al., N.D.).
[c] (Allinson et al., 2007; Antonopoulou et al., 2008; Baskaran et al., 2003; Britz et al., 2004; Carvalho et al., 2013; Chen et al., 2018; Durham and Hourigan, 2009; Erguder et al., 2001; Hassan and Nelson, 2012; Ince, 1998; Kushwaha et al., 2011; Labatut et al., 2011; Liu and Haynes, 2011; Mainardis et al., 2019; Nadais et al., 2010; Prasad, 2006; Vivekanand et al., 2018; Wilkinson et al., 2007).
[d] (Astals et al., 2015; Gopalan et al., 2013a; McGahan et al., 2016; Skerman et al., 2016; Tucker, 2018).
[e] More typically within narrower range of 330–360 $m^3_N\ CH_4 \cdot t^{-1}$ VS_{added} (Skerman et al., 2016); [&]For irrigation pond effluent.
[f] (Jensen et al., 2014, 2016; Liu and Haynes, 2011; McCabe et al., 2014, 2020; Ridoutt et al., 2015; Schmidt et al., 2019; White et al., 2013).
[g] based on data in Tables 1 and 3 of Jensen et al. (2014); "-" means no Australian data reported in the available literature.

Table 2
Typical composition of various agro-industrial organic solid waste types, given as a range of reported values or as individual values where only single values were found.

Characteristic	Spent straw-based piggery litter[a]	Scraped beef feedlot manure[b]	Red meat processing[c]	
			Paunch contents	Screenings/sludge/float from primary effluent treatment
TS (%)	17–84.6	19.6–95.6	11.7–18.1	14.6–36
VS:TS ratio	0.68–0.93	0.64–0.77	0.91–0.95	0.76–0.98
Total COD	1.3–1.4 g COD.g^{-1} VS	235–455 g kg^{-1}	106–111 g kg^{-1}	205-1,053 g kg^{-1}
VFAs ($g \cdot kg^{-1}$ TS)	0.9–114	–	5.5–15.4	1.4–44
TN or TKN ($g \cdot kg^{-1}$ TS)	2–13	9.5–41	5.1–17.9	3.3
Total P ($g \cdot kg^{-1}$ TS)	2–25	7.5–12.1	1.7–6	0.8
Total K ($g \cdot kg^{-1}$ TS)	6–28	7.3–29.2	–	–
Total S ($g \cdot kg^{-1}$ TS)	1–7	3.1–6.4	–	–
pH	5.7–8.5	8.4	–	4.16[c]
B_0 ($m^3_N\ CH_4 \cdot t^{-1}\ VS_{added}$)	140–270	70–280	237–254	262–912

The range of values given represent data reported by
"-" means no Australian data reported in the available literature. *Low pH may be atypical, and would depend on extent of fermentation and use of chemicals during primary treatment.
[a] (Tait et al., 2009; Tucker, 2018; Yap et al., 2016, 2017);
[b] (Gopalan et al., 2013b; Pratt et al., 2014; Tucker et al., 2015; Watts and McCabe, 2015), Data from (Gopalan et al., 2013b) is for pad manure.
[c] (Astals et al., 2014; Harris et al., 2017; Jensen et al., 2016; McCabe et al., 2020);

systems) and some using dairy or poultry manure (IEA Bioenergy Task 37, 2020). Targeted previous studies have explored biogas production for organic waste from Australian agricultural industries such as solid organic waste (Tait et al., 2009) and liquid organic waste (Gopalan et al., 2013a) from pork production, beef feedlot manure (Gopalan et al., 2013b), sugarcane trash (Janke et al., 2019), and red meat processing wastewater (Jensen et al., 2014). However, to date, there has not been a consolidated review of key information gaps pertaining to aggregated AcoD opportunities. This affects the understanding of and ability to harness such opportunities.

To facilitate development of future biogas projects, the current paper provides a critical evaluation of important information on Australian dairy, pork, intensive beef and red meat processing sectors, their energy demand, organic waste and biogas potential, potential AD approaches, and important opportunities and constraints for aggregated AcoD.

2. Method for selection of literature for review

The public and peer-reviewed literature were searched for information and past initiatives on mapping and characterization of agro-industry organic waste from the Australian pork, beef, meat processing and dairy sectors. For public documents, the review used only reputable sources such as:

1. Environmental codes of practice and guidelines frequently called up in environmental legislation and/or used by environmental regulators in Australia;

2. Benchmark performance metrics, statistics, data and guidance documents on respective agro-industries as published by national representative bodies of those industries (e.g. Australian Pork Limited; Dairy Australia; Meat and Livestock Australia; Australian Meat Processor Corporation); and
3. Statistics published by the Australian federal government.

Anaerobic treatment options for Australian agro-industry organic waste have been researched to a limited extent by Australian laboratory or pilot-scale studies (See Section 3), and where peer-reviewed literature were available on this they were included. The review also drew from peer-reviewed published literature on full-scale anaerobic treatment options, which included sources from Australia where available, as well as international experiences that broadly aligned with Australian conditions (e.g. remote agro-industries; large spatial footprint often available; uncertain water supply, See Section 3).

3. Australian agro-industries, organic waste and anaerobic treatment options

3.1. Australian agro-industries overview

This section provides an overview of the dairy, pork and beef sectors (including red meat processing (RMP)). These agricultural industries are important to the Australian economy as follows:

- The dairy sector is Australia's 4th largest rural industry with a farmgate production value of AUD4.4 billion in 2018/19 (Dairy Australia, 2021);
- Australian pork is an important food protein source, with a gross production value-add of ~AUD1.4 billion (Acil Allen Consulting, 2017); and
- Australia's beef industry has an off-farm meat value (including domestic plus export) of approximately AUD19.6 billion in 2018/19 (Meat and Livestock Australia, 2019a), with Australia producing approximately 3% of the world's beef and being the third largest beef exporter (Meat and Livestock Australia, 2019a).

Total production for dairy and red meat sectors (including beef feedlots and RMP) are geographically unevenly distributed across Australia (Fig. 1).

Specifically, the majority of dairy is produced in Victoria (VIC), the majority of intensive beef production occurs in Queensland (QLD), and sheep production occurs mostly in the southern states of New South Wales (NSW), South Australia (SA) and VIC and in the southern parts of Western Australia (WA) (Fig. 1). This has implications for availability of organic waste as potential feedstocks for AD or aggregated AcoD. In contrast, pork production is fairly evenly distributed across QLD, VIC, SA, NSW and WA (Fig. 1).

3.2. Dairy on-farm – organic waste types and anaerobic processing options

Australian dairy production is predominantly pasture-based, albeit that a significant proportion of the industry also uses intensive feeding systems (Watson et al., 2015) which can improve performance and increase climate resilience (used by ~26% of total production) (Dairy Australia, 2017). This has implications for waste availability as described in Section 4. Dairy manure is the main organic waste being produced, predominantly collected as a liquid effluent (Fig. 2) and sometimes as a dry-scraped semi-solid (Birchall et al., 2008). Dairy effluent consists of wash water mixed with cattle urine and dung, cleaning chemicals, spilt feed and bedding if used (Birchall et al., 2008). Australian dairy effluent is typically dilute, with a low solids content (Table 1).

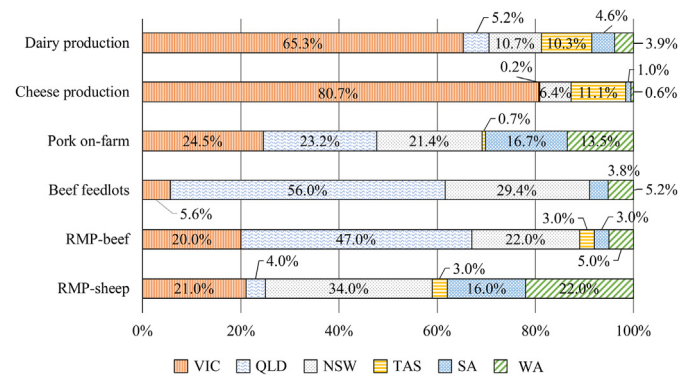

Fig. 1. Distribution of total primary production across various Australian states, proportioned by herd numbers for dairy production (Dairy Australia, 2019a), cheese tonnages for cheese production (Dairy Australia, 2019b), sow numbers for pork on-farm (Acil Allen Consulting, 2017), herd capacity for beef feedlots (Graham, 2020), and beef carcass weight or sheep numbers for RMP (Meat and Livestock Australia, 2020).

Organic matter in dairy effluent is expected to be predominantly particulate. For example, the solid fraction contributed approximately 2/3rd of the methane yield in dairy manure when separated (Rico et al., 2012). The rate of AD of a particulate waste type is likely limited by hydrolysis (Batstone and Jensen, 2011). Continuously stirred heated tank digesters (CSHTDs) could be considered but is likely to be limited hydraulically with dilute liquid organic waste (Batstone and Jensen, 2011) such as dairy effluent. Water is a carrier for the cleaning of dairy manure from milking yards and milking sheds. As such, water savings initiatives could be implemented, but only to the extent that maintains animal health and milk quality. Instead, separation processes could be considered to recover manure solids from the dilute dairy effluent (Hjorth et al., 2010), which increases the solids content by forming a recovered solids fraction. The pre-concentrating of solids could help overcome hydraulic limitations of CSHTDs (Batstone and Jensen, 2011). CSHTDs could offer flexibility and choice of other waste types to be co-digested together with dairy manure, thereby enhancing biogas production and providing a more balanced feedstock composition for improved digestion performance (See Section 5.1).

Alternatively, dairy effluent could be treated in covered anaerobic ponds (CAPs) if adequate footprint is available and site conditions are appropriate for their construction. Uncovered effluent ponds are commonly used in many Australian dairy farms (Watson et al., 2015), therefore covering of effluent ponds for biogas capture is an incremental change from current practice. Anaerobic ponds can be relatively cost effective to construct (Batstone and Jensen, 2011). Also, prolonged retention times could facilitate conversion into biogas (Heubeck and Craggs, 2010). However, anaerobic ponds are intolerant of floating organic waste which form excessive scum or crust layers which can damage a cover (Jensen et al., 2015) and would be inaccessible once an effluent pond is covered (Birchall et al., 2008). Moreover, anaerobic ponds may offer minimal ability to control (e.g. temperature) and large volumes can make process corrections expensive (Batstone and Jensen, 2011).

3.3. Dairy processing – organic waste types and anaerobic treatment options

Combined liquid effluent and whey are the main organic waste types from dairy processing, and both are liquid waste streams. The combined liquid effluent is produced from dairy processing and equipment cleaning and is typically treated onsite to varied extents,

Fig. 2. Photos of typical manure management in Australian dairy, pork and beef feedlot sectors, showing: (A1/A2) dairy yard wash systems producing liquid dairy effluent, Source: Dr. Scott Birchall and Dairy Australia; (B1/B2) conventional piggeries producing liquid effluent, Source: Dr. Stephan Tait and Pork CRC; (C1/C2) deep litter piggeries producing stockpiled spent litter, Source: Dr. Stephan Tait and Pork CRC; (McCabe et al., 2019); and manure harvested at beef feedlots using (D1) a box scraper that maintains a manure interface layer and collects a "cleaner" manure, Source: Mr. Peter Watts or (D2) a front-end loader producing a rough finish and a manure that is significantly contaminated with clay or aggregate, Source: Mr. Peter Watts.

including by flow-balancing/equalization, physico-chemical, biological and tertiary treatment (Britz et al., 2004). Whey is produced predominantly from cheese making, and to a lesser extent from yoghurt making (Arcadis, 2019). Whey consists of water and milk solids not retained in the curd, including most of the lactose and some fat and soluble protein (Prazeres et al., 2012). Three main types of whey are produced with distinct characteristics, namely: acid whey (pH < 5); sweet whey (pH $= 6-7$); and salty whey (Prasad et al., 2005). Globally, whey has been a major environmental challenge for dairy processing (Fernandez-Gutierrez et al., 2017), containing over 50% of the milk solids including 20% of the proteins, and most of the lactose (Nadais et al., 2010). Generally, useable products can be recovered (e.g. food-grade whey powder, or whey for infant formula, biscuits and ice-cream (Dairy Australia, 2019b)). However, where the recovery of such products is not economical, whey is instead used in piggeries as an animal feed, is irrigated onto farmland or is disposed to sewer (Hauser, 2017).

Dairy processing also produces solid organic waste, including reject or unsold product, and sludges from processing and onsite effluent treatment (Wilkinson et al., 2007). No reliable direct data could be found on amounts of these waste types but amounts currently unutilized are expected to be minimal. Dairy processors in VIC most commonly "dispose" these waste streams as stockfeed in piggeries, or to a lesser extent for off-site composting or application to agricultural land as a soil ameliorant/fertiliser (Allinson et al., 2007). Consequently, aggregated AcoD opportunities could be limited for such waste streams, which were therefore excluded from further consideration in the current work.

Anaerobic lagoons are the most used system world-wide for treatment of liquid effluent from dairy processing (Nadais et al., 2010), except where space is highly constrained and not able to accommodate their typically large footprint (e.g. in Europe). However, high-rate anaerobic processes have also been widely used in practice, with various system designs providing retention of biomass to minimise washout (Nadais et al., 2010) and allow some extent of process intensification. Liquid effluent volumes are

typically high, ranging between 0.96 and 2.43 L L^{-1} of milk processed (Prasad et al., 2005), and hydraulic loading issues with CSHTDs limit its application in full-scale treatment (Britz et al., 2004). Granular sludge or carrier packings can be used in high-rate systems, and biomass and fat retention can be achieved via filters, membranes, or other types of solid separators (Nadais et al., 2010). Due to hot water used for cleaning, an elevated temperature of the combined dairy processing effluent (Nadais et al., 2010) which could facilitate anaerobic treatment (Kushwaha et al., 2011). In general, ammonia inhibition risk seems relatively low with dairy processing effluent, however, if substantial protein mineralization did occur, this could result in levels of ammonia that are toxic to AD (Nadais et al., 2010). Milk protein is the only significant nitrogen source in dairy processing effluent, so if pretreatment of effluent was to remove the protein (e.g. by coagulation and/or precipitation), this could cause nutrient deficiency of subsequent biological treatment (Nadais et al., 2010). Precipitation of milk protein further produces aggregates of solid material which are poorly bioavailable/difficult to degrade (Nadais et al., 2010).

Whey is more concentrated than combined dairy processing effluent, and a low alkalinity can pose significant stability and control challenges for high-rate anaerobic processes by causing low sludge settleability and biomass wash-out and a risk of pH depression by rapid fermentation of lactose (Nadais et al., 2010). Methanogenesis is susceptible to low pH inhibition (Nadais et al., 2010). A high salt content (e.g. in salty whey) could also be inhibitory to AD processes (Nghiem et al., 2017).

Sludge floatation, biomass loss, and long chain fatty acids (LCFA) inhibition are significant concerns in AD of dairy processing waste (Nadais et al., 2010). A potential strategy to overcome this might be to seed a digester with a resilient anaerobic consortia (Nadais et al., 2010), albeit that such a microbial consortia must have good settleability (Nadais et al., 2010), or improved anaerobic reactor designs should facilitate biomass retention (Nadais et al., 2010). To overcome limitations with excessive fermentation and pH depression during AD, rapid pre-acidification of liquid effluent or whey could be separated from subsequent methanogenesis in a two-stage AD process (Nadais et al., 2010), otherwise requirements for external alkalinity could be cost prohibitive (Nadais et al., 2010).

Anaerobic treatment has to date been applied in Australian dairy processing, but it appears only by a small number of larger processors. A list of example installations is provided by GHD (2017). Biogas energy potential of combined liquid effluent and whey appears to be significant, as does the energy demand of dairy processing (See Section 4). Moreover, dairy processing also has a significant need for natural gas (See Section 4) which is readily displaced with biogas, so more dairy processors in Australia are likely to adopt biogas systems into the future.

3.4. Pork production on-farm – organic waste types and anaerobic digestion options

Manure is the main organic waste type from on-farm pork production. About 90% of Australia's pig herd is housed indoors (Tucker, 2018). The remainder 10% is reared outdoors (Tucker, 2018) for which manure is not collectable and is thus not further considered in this work. Indoor housing types include conventional sheds for which manure is collected as a liquid effluent or deep litter sheds for which manure is collected as spent bedding (also termed spent piggery litter) (Tucker, 2018) (Fig. 2). An estimated 20% of the national Australian pig herd is produced in a combination of conventional housing and deep litter housing (data not shown), meaning that both piggery effluent and deep litter are potentially available for AD.

Piggery effluent is a dilute mixture of manure, urine, spilt feed and wash water (Tucker, 2015) (Table 1). Organic matter in piggery effluent is predominantly particulate (Tucker, 2015). The rate of AD of a particulate waste type is likely limited by hydrolysis (Batstone and Jensen, 2011). However, piggery effluent can also have a significant dissolved organic matter content (Table 1), predominantly comprised of volatile fatty acids (Gopalan et al., 2013a) which would be readily converted into biogas. Piggery effluent is usually treated onsite in effluent ponds and is often recycled as flush water and/or irrigated onto agricultural land to offset fertiliser use (Tucker, 2015).

Biogas can be produced from piggery effluent in CAPs (Tucker, 2015). For example, as at 2018, about 13.5% of total Australian pork production had adopted biogas systems (Skerman et al., 2018) and Skerman and Tait (2018) identified that most of these systems were using CAPs. Because methane from piggery effluent ponds has been identified as the dominant GHG source across the Australian pork supply chain (Wiedemann et al., 2016), Australian government Emissions Reduction Fund (ERF) legislation (Commonwealth of Australia, 2019) has to date financially incentivized the capture and combustion of manure methane. Up to the year 2020, this had enabled Australian pork farmers to abate an estimated 664,800 tonnes (t) of CO_2 equivalent of manure management emissions (Australian Clean Energy Regulator, 2020). Fig. 3 shows examples of CAP installations at Australian piggeries.

Some Australian pig farms instead use mixed heated digesters, specifically in-ground mixed heated CAPs and mixed tank digester systems (Skerman and Tait, 2018). Such mixed heated digesters have significantly higher capital costs and increased operational complexity compared to CAPs (Skerman and Tait, 2018). CAPs on the other hand are typically subject to seasonal fluctuations in biogas production, with lower biogas production occurring in cooler months and higher biogas production in warmer months, as shown for monitored case study sites in Victoria (Birchall, 2010) and Queensland (Skerman et al., 2011). However, Australia has a relatively temperate climate, and particulate organic matter can settle out in CAPs, thereby increasing solids retention time. In this way, the effect of lower operating temperature may be buffered to some extent by the longer solids retention time (Heubeck and Craggs, 2010) allowing more time for conversion into biogas. Mixed heated digesters instead speed up the rate of hydrolysis and conversion into biogas by operating at a consistent higher temperature.

Spent piggery litter consists of pig faeces, urine and some spilt feed, mixed with an absorbent bedding material of rice husks, wheat straw, barley straw or saw dust (Tucker, 2015). The type of bedding used typically depends on availability and cost. The bedding is progressively added whilst a group of pigs grow to ensure that the shed areas remain dry (Kruger et al., 2006). When the group of pigs leave the shed at the end of their growth cycle, the spent piggery litter is removed as a batch of material (Tucker, 2018). This affects anaerobic treatment options as discussed further below. Unlike piggery effluent, spent piggery litter is a stackable solid organic waste type. Spent bedding has widely varying properties depending on bedding type and extent of soilage by the pigs (Tait et al., 2009).

To the authors' knowledge, there are currently no anaerobic digesters in Australia operating with spent piggery litter as feedstock. The batch-wise production of spent piggery litter would pose unique challenges, because spent litter would not be continuously available for digestion unless stored onsite. Such storage may lead to unwanted organic matter losses and subsequent methane yield losses. Some types of spent piggery litter such as litter on rice husks and saw dust are not attractive for AD, because of poor biogas potential (e.g. rice husk spent piggery litter (Tait et al., 2009)). Accordingly, only straw-based spent piggery litter was further

Fig. 3. Photos of typical covered pond installations and biogas electricity generator with hot water recovery at conventional Australian piggeries. Source: Dr. Stephan Tait and Pork CRC.

considered in the current work, having a reasonable biodegradability (e.g. 26–58% (Tait et al., 2009)) and reasonable hydrolysis rate (e.g. first-order rate as high as 0.09 d^{-1} (Tait et al., 2009)). Inshed pre-conditioning (e.g. chewing, trampling, pre-fermenting and mixing with manure) may enhance the anaerobic biodegradability of litter by 25% or more as compared to raw straw (Tait et al., 2009).

Because spent piggery litter is a solid organic waste type, dry AD may be an attractive option. Dry AD can have smaller reactor volumes (Li et al., 2011), lower energy requirements for heating and less parasitic energy loss (Li et al., 2011), and materials handling can be simpler than with slurries in liquid AD (Li et al., 2011). Although continuous solid-state AD has been popular for municipal solid waste, batch digestion may be more suitable for ligno-cellulosic waste such as spent piggery litter. Specifically, batch solid-state digestion is relatively simple and has minimal maintenance requirements and a comparatively low capital cost (Li et al., 2011). Batch digestion may also be more suitable for spent piggery litter because of its typical batch-wise production. With batch solid-state AD, the feedstock is digested in a gas-tight container or room, at 30%–40% dry matter (Li et al., 2011). The digestion of a new batch is inoculated with digested material or water leachate from a previous completed batch (Li et al., 2011). Leachate can be percolated over the batch of material which can increase mass transfer and promote the efficiency of contact between microorganisms and organic matter (Meng et al., 2019). This system is typically called a leachbed. A previously Australian study tested a leachbed for spent piggery litter at small pilot scale (Yap et al., 2016) and a larger farm-scale study in Italy tested a leachbed with AD of rice straw and piggery wastewater (Mussoline et al., 2014). In the case where Australian piggeries have both conventional and deep litter housing onsite, it may be possible to integrate a leachbed system together with a CAP at the same piggery (Fig. 4).

This can use a leachbed to process the spent piggery litter and a CAP to process the piggery effluent and to be (together with a secondary effluent pond) a source of leachate for the leachbed. The CAP would also treat leachate from the leachbed for further biogas recovery. Future studies could also explore ensiling methods for spent piggery litter, similar to what has been previously tested for sugar-cane waste to preserve methane yield during storage (Janke et al., 2019). Ensiling could then make spent piggery litter available for continuous AD options.

3.5. Beef feedlots – waste type and anaerobic digestion options

With beef feedlots, cattle manure is the main organic waste type (Tucker et al., 2015). Beef feedlot manure is typically dry-scraped as a semi-solid or solid (Table 2) and stockpiled to decompose and passively or actively composted (Bai et al., 2020) before land spreading (Gopalan et al., 2013b). Frequent pen cleaning has benefits of promoting pen drying and minimizing odour emissions as compared to wet pens (Tucker et al., 2015). However, feedlot pens are infrequently cleaned at intervals of 3–6 months (Watts and McCabe, 2015) albeit that cleaning at intervals of 13 weeks is typically recommended (Tucker et al., 2015). Some equipment used for manure scraping harvests manure down to the soil and gravel underlay of the feedlot pen (e.g. wheel loaders), which contaminates manure with soil and gravel (Tucker et al., 2015) (Fig. 2). Other equipment scrapes with good depth control and a smooth pen finish (e.g. graders), maintaining a manure interface layer and harvesting a "cleaner" manure (Tucker et al., 2015) (Fig. 2). A cleaner harvested manure would likely facilitate trouble-free AD.

To the authors' knowledge, there are currently no anaerobic digesters in Australia operating with beef feedlot manure as feedstock; however, there has been on-going interest from Australian beef feedlots to explore AD options. The frequency of pen cleaning is important because pen manure rapidly decomposes on the feedlot pen surface. As much as 60–70% of the VS can be lost over a 20-day period (Davis et al., 2012) and this can result in a less biodegradable manure with lower B_0 (decreasing from 230 to 360 $m^3_N\ CH_4 \cdot t^{-1}\ VS_{added}$ for fresh manure (Gopalan et al., 2013b) down to 70–280 $m^3_N\ CH_4 \cdot t^{-1}\ VS_{added}$ for manure aged 3–8 weeks (Gopalan et al., 2013b)).

Different extents of coarse grit and stone removal would likely be needed to prevent inerts such as gravel from entering a digester,

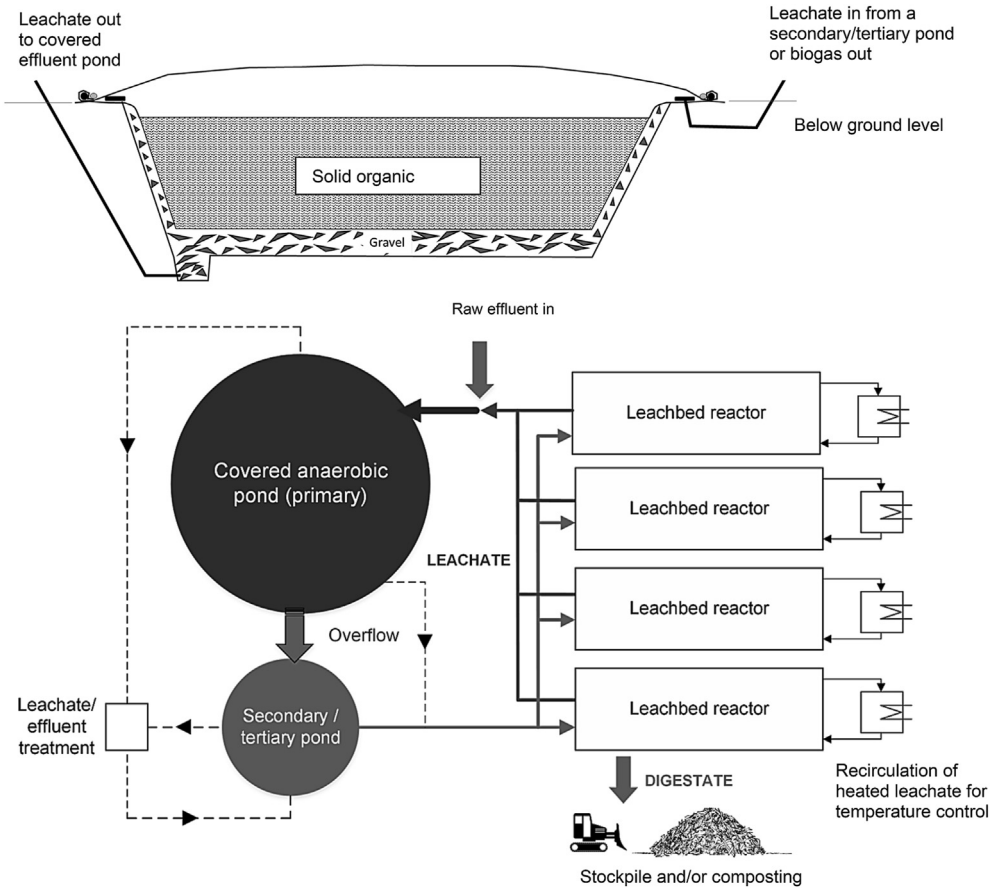

Fig. 4. A conceptual diagram of a potential approach to integrate a leachbed and a covered anaerobic pond to simultaneously utilise the solid organic waste and liquid organic waste at a piggery. Adapted from Yap (2017).

to prevent damage to AD plant components (Watts and McCabe, 2015) and reserving useful digester volume for AD of manure. Beef feedlot manure is a predominantly particulate substrate (Table 2); the rate of AD of particulate organic waste is likely limited by hydrolysis (Batstone and Jensen, 2011). Feedlot manure exhibits low to moderate hydrolysis rates (Gopalan et al., 2013b). A significant quantity of water would be required to prepare feedlot manure as an AD feedstock. It may be possible for the liquid fraction of digestate to be recycled for this purpose (Fig. 5), albeit that ammonia levels would need to be monitored to prevent build up to inhibitory levels.

A leachbed approach has been previously trialed by Colorado State University (Watts and McCabe, 2015), which used a top-layer of sand to prevent clogging and to promote leachate hydraulics (Watts and McCabe, 2015). It may be possible to instead mix beef feedlot manure with a crop residue for digestion in a leachbed (as in Section 3.4). A crop residue could act as a bulking agent to reduce clogging and promote leachate flow and could provide additional biogas production if reasonably biodegradable (e.g. straw). In the same leachbed study of Colorado State University, leachate was passed through the leachbed system and not recirculated, and could have flushed essential nutrients from the leachbed (Watts and McCabe, 2015). This could have caused an observed increase in COD removal in the same Colorado State University trial when an external source of nutrients was dosed (Watts and McCabe, 2015).

A potential future approach to facilitate practical and cost-effective AD could be to only harvest manure from selected pens at a beef feedlot, thereby collecting the amount of manure to satisfy onsite demand for biogas energy (See Section 4). The selected pens could be converted to concrete flooring (albeit at considerably higher capital expense) to minimise contamination of manure with gravel or soil and allow use of liquid flushing systems as in dairies and piggeries. The manure slurry produced may then be suitable for low-cost CAP digestion, albeit that liquid digestion requires significant quantities of water. A similar approach was previously proposed by McCabe and McMeniman (2017), except where feedlot manure was frequently dry-scraped, screened of grit and large particulates (e.g. a mesh size of 10 mm to remove gravel), and slurried up in a mixing tank for subsequent digestion in a CAP (Fig. 5).

It may be possible to reduce the overall water demand by recycling effluent for flushing of feedlot pens, but nutrient levels in the recycled effluent would need to be carefully monitored to ensure that they do not accumulate to inhibitory levels for AD. Any future AD of feedlot manure would likely need separation of hair, plastics and other sundry contaminants (e.g. baling twine) (Watts and McCabe, 2015). The few beef feedlot installations in the USA and Canada inspected in the study of Watts and McCabe (2015) all had reported significant challenges with such inert particulate contaminants, also including sand from scraping of concrete floors. AD strategies could be piloted at Australian beef feedlots to clarify specific integration opportunities and challenges.

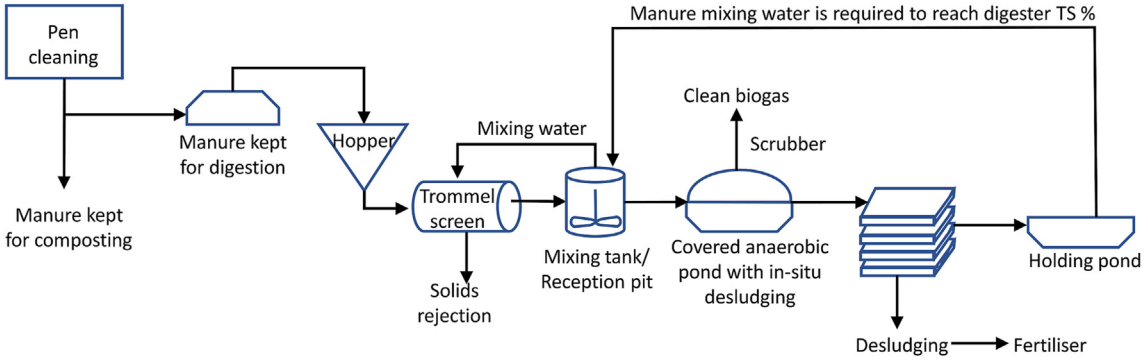

Fig. 5. Conceptual diagram of a potential approach to harvest and anaerobically digest beef feedlot manure. Adapted from (McCabe and McMeniman, 2017).

3.6. Red meat processing – organic waste and anaerobic treatment options

Australian RMP follows a well-versed chain of activities that produces a range of organic solid and liquid waste types (McCabe et al., 2019). In listed order, the typical processing steps producing organic waste include (Australian Meat Processors Corporation, 2010): preparation for slaughter (including livestock unloading and holding); slaughter (including stunning, hide removal, evisceration, trimming and washing); offal processing; chilling (only produces liquid effluent from cleaning); boning; and rendering of by-products (at a number of facilities) to produce tallow and meal.

Liquid effluent is produced from sterilisation, rinsing, washing, cleaning and sanitizing to strict hygiene standards (Liu and Haynes, 2011). This liquid effluent is typically treated onsite to remove FOG, nutrients, organic matter and suspended solids before being irrigated onto agricultural land or disposed to sewer (Liu and Haynes, 2011). Past research has classified effluent streams from different processing areas according to their distinct characteristics to identify tailored anaerobic treatment options (Jensen et al., 2014). By this classification, RMP effluent comprises a red stream (from slaughter floor and rendering), a green stream (from offal processing and paunch handling) and a separate high-volume dilute effluent sub-stream (from boning and cattle wash) (Jensen et al., 2014).

Liquid effluent streams from different processing areas are typically transported and treated separately within a RMP facility before being combined for pond-based treatment (Jensen et al., 2014). Pond-based treatment has been common where adequate land is available for a large footprint, because it can be relatively low-cost, simple (Liu and Haynes, 2011) and effective at reducing organic matter loads (Mittal, 2006). However, where facilities are in urban areas, a shortage of land and risk of odour can make effluent ponds unsuitable (Liu and Haynes, 2011). The composition of liquid effluent at a number of Australian RMP facilities (Table 1) is influenced by rendering onsite (McCabe et al., 2020), resulting in larger volumes with higher organic loading, elevated FOG levels and generally warmer temperatures (as high as 50–60 °C) (Johns, 2012). Higher temperatures emulsify FOG into effluent (Johns, 2012). Moreover, poor gas solubility at elevated temperature may reduce the efficacy of FOG removal during primary treatment (Jensen et al., 2014). The efficiency of primary treatment influences subsequent AD as further discussed in this section below.

Some large Australian RMP facilities have implemented CAPs to capture offensive odour and utilise biogas onsite as a boiler fuel (McCabe et al., 2020). This has indicated a baseline feasibility for biogas recovery from combined RMP effluent in Australia. Hot streams may offer heating opportunities for anaerobic processes (McCabe et al., 2020), but temperatures may be initially too high for biological processes (Johns, 2012). Progressive cooling causes coagulation and phase separation of fats (Banks and Wang, 2004) and may limit indirect heat recovery options using heat exchangers. Effective primary treatment is important upstream of an AD system, as FOG can cause fouling and accumulation in infrastructure (McCabe et al., 2020), and FOG and RMP solids can form a recalcitrant crust in CAPs that may damage a cover (Jensen et al., 2015) and/or reduce treatment efficacy (McCabe et al., 2014).

If considered separately, the red stream has a solids concentration generally too low for a CSHTD (Jensen et al., 2014) but could be amenable to high-rate anaerobic treatment systems that are tolerant of FOG (Jensen et al., 2014). The high nitrogen content of red stream may pose an ammonia inhibition risk and should be carefully monitored (Jensen et al., 2014). Conventional high-rate systems such as up-flow anaerobic sludge blanket reactors (UASB) have been applied to RMP effluent at laboratory and pilot scale (Banks and Wang, 2004), but have shown poor tolerance of solids, especially FOG (Jensen et al., 2014). High FOG content may make it difficult to form stable granules and can exacerbate sludge losses (Banks and Wang, 2004). Instead, FOG-tolerant high-rate options are required. Jensen et al. (2014) previously mentioned anaerobic membrane bioreactors (AnMBRs) or anaerobic floatation reactors as prospective technology options. AnMBRs could achieve a high-quality treated effluent and biomass retention because of the membrane separation (Jensen et al., 2015). A previous Australian study (Jensen et al., 2015) demonstrated the stable operation of a pilot-scale AnMBR treating red streams of two RMP facilities, provided that membrane fouling could be managed by maintaining a minimum in-reactor active biomass level and not exceeding a total operating solids level of 40 g L^{-1}. As a result, an AnMBR should not operate above a particular feed solids loading limit (Jensen et al., 2015). AnMBRs are commercially available but have only been explored at laboratory and pilot-scale in Australia, and further research is required to consider their feasibility in RMP at larger scale.

Organic solid waste produced by Australian RMP include paunch contents, manure and yard waste, and screenings/float/sludge from liquid effluent treatment (Ridoutt et al., 2015). Early screenings and FOG float from primary treatment may be recycled to rendering if an appropriate quality can be maintained, and this would decrease final solid waste amounts. However, whether it is profitable to recycle screenings and FOG float would heavily depend on the relative value of second grade tallow as compared to the value of organic matter retained in combined liquid effluent for biogas energy recovery (Fredheim, 2018).

Paunch contents is a major solid waste type from RMP produced when edible offal products (e.g. paunch and runners) are emptied

and washed before further processing (Australian Meat Processors Corporation, 2010). Paunch contents are typically washed into the effluent (Australian Meat Processors Corporation, 2010) producing the green stream with relatively poor biodegradability compared to the red stream. Consequently, suspended solids in the green stream are often removed to minimise accumulation of inerts in downstream effluent ponds (Johns, 2012). For effluent treatment in a CAP, green stream solids would likewise need to be removed to prevent accumulation of inerts under a pond cover.

CSHTDs may be an option for screened green stream solids. However, a low hydrolysis rate and low methane yield of paunch contents and intestinal fecal material have somewhat limited the economic return of AD of these waste types to date (Banks and Wang, 2004). It could be beneficial to co-digest red stream, green stream solids and fat-rich sludge together in a single digester. This could boost biogas production via increased biodegradable organic loading, reduced LCFA inhibition by dilution, and potential synergistic effects on the microbial community (Astals et al., 2014). Australian research has also explored low-intensity thermophilic pretreatment of screened paunch solids to improve its conversion into biogas (Jensen et al., 2016). The research indicated that this low-intensity thermophilic pretreatment did not change the rate nor the extent of digestion but enabled significant process intensification at pilot-scale (Jensen et al., 2016). This process intensification could have benefits due to more efficient use of digestion capacity at full-scale.

4. Organic waste availability and biogas potential

Organic waste amounts and biogas potential were estimated in the current work. The method used for biogas potential estimation was similar for each agro-industry and is outlined in Table 3. Waste availability was estimated as follows:

Dairy on-farm: Manure amounts were estimated by Australian National Greenhouse Accounts methods (Commonwealth of Australia, 2018), assuming an average milk yield of 16.5 kg·cow^{-1}·d^{-1}, liveweight gain of milking cows at 0.016 kg d^{-1}, 550 kg milking cow weights, and other default factors (Commonwealth of Australia, 2018). Accordingly, daily manure output was estimated at 4.5 kg volatile solids (VS) per head per day, marginally lower than that reported by Birchall et al. (2008) using the method of Nennich et al. (2005). This daily manure output was multiplied by 1.44 million cows (Dairy Australia, 2018a) and a 0.2 proportion not voided on pastures (was between 79% and 82% in 2018 (Christie et al., 2018)), giving the estimated collectable manure amount in Table 3.

Dairy processing: Liquid effluent amounts were estimated assuming 1.7 L of effluent produced per L milk processed (Mehta et al., 2016). When multiplied by total milk production of 8.8 GL·annum^{-1} (Dairy Australia, 2018a), this gave the estimated liquid effluent amount in Table 3. Whey production ranges from 8.12 L whey·kg^{-1} of cheese manufactured (Hauser, 2017) to 10 L whey·kg^{-1} of cheese manufactured (Arcadis, 2019). A nominal value of 9 L whey·kg^{-1} of cheese was assumed, which when multiplied by 378 kt·annum^{-1} of cheese manufactured in Australia in the year 2017/18 (Dairy Australia, 2019b) gave 3.4 GL·annum^{-1} of whey produced nationally. An estimated 77% of this whey would be converted into dried products (Arcadis, 2019), and the remainder proportion (Table 3) was assumed to be potentially available for value-adding via AD or AcoD.

Pork on-farm: The Standard Pig Unit (SPU) approach was used, with an SPU being a quantifier of equivalent pig manure production including feed wastage (Tucker, 2018). The equivalent number of SPUs corresponding to a single breeding sow at a typical Australian piggery is 10.731 (Tucker, 2018), quantifying her own manure output and that of all her progeny and other pigs associated with her production. The total number of breeding sows across Australia has been estimated at 277,000 (Acil Allen Consulting, 2017), corresponding to an estimated total herd equivalent of 2,972,500 SPUs. For conventional piggeries (i.e. housed indoors with liquid effluent), one SPU is equivalent to a 90 kg VS·annum^{-1} manure organic matter output (Tucker, 2018). The proportion of the total industry in conventional sheds is about 70% (Commonwealth of Australia, 2018). Accordingly, manure organic matter in piggery effluent was estimated and is given in Table 3. For deep litter piggeries, one SPU corresponds to a spent litter output of 320 kg dry matter·SPU^{-1}·annum^{-1} (Kruger et al., 2006) with estimated organic matter of 256 kg VS·SPU^{-1}·annum^{-1} based on the VS percentage reported by Tait et al. (2009). The higher VS output (as compared to conventional piggeries) is mostly due to the additional organic matter of the bedding material. It was assumed that 60% of all deep litter systems uses straw. This is a coarse approximation because no data were available on the relative use of various litter types. The remainder spent litter systems, which use other litter types that are not attractive for biogas due to low biomethane yield (See Section 3.2), were excluded from further consideration in the current work. When 256 kg VS·SPU^{-1}·annum^{-1} of spent litter output is multiplied by 2,972,500 SPUs, multiplied by the proportion of industry with deep litter systems which is estimated at 20% (Commonwealth of Australia, 2018), and finally multiplied by the proportion assumed to use straw type bedding (i.e. 60%, see above), the total spent straw-based piggery litter output was equivalent to that given in Table 3.

Beef feedlots: The standard cattle unit (SCU) approach was used, which quantifies manure output relative to that of an animal with 600 kg live weight (Tucker et al., 2015). A table of SCU factors vs. number of head of cattle at various live weights is provided by Meat and Livestock Australia (2012). The Australian National Inventory Report (Commonwealth of Australia, 2018) classifies Australian feedlot cattle and provides estimated numbers of cattle (years 1990–2018) in three main classes, namely domestic market feedlot cattle ("Domestic (70–80 days"), export market feedlot cattle with a middle finish age ("mid-fed (80–200 days") and export market feedlot cattle with an extended finish age ("long-fed (200+ days)"), and states average liveweights for each of these age classes. These were used to determine the equivalent SCU factor for each age class, which were then multiplied by respective cattle numbers in 2018 (Commonwealth of Australia, 2018) to provide an estimated total SCU in Australian cattle feedlots of 928,828 in 2018. Equivalent manure output was assumed at 420 kg dry matter·SCU^{-1}·annum^{-1} with a 70% VS proportion based on manure characteristics reported by Tucker et al. (2015) and corresponding to a manure being harvested fairly clean (See Section 3.5) with some decomposition losses on the feedlot pen (e.g. 20 days on the pen (Tucker et al., 2015)). Note that manure production could be significantly higher than 420 kg dry matter·SCU^{-1}·annum^{-1}, so estimates from the current work may be conservative. If bedding (e.g. wood chips) is used in a feedlot, this would add to manure mass harvested (Tucker et al., 2015), but was excluded in the current calculations because of expected poor biodegradability. Accordingly, estimated manure output amount was as given in Table 3.

Red meat processing: In 2019, total annual Australian beef and veal production was estimated at 2.4 mega tonnes (Mt) carcass weight (Meat and Livestock Australia, 2020) and total sheep and mutton production at an additional 0.7 Mt·annum^{-1} carcass weight (Meat and Livestock Australia, 2020). Goat meat was excluded from the current calculations, being a relatively minor contributor to total processing based in proportions given by Meat and Livestock Australia (2019b). Combined liquid effluent volumes from Australian RMP is reported to be (on average) 8.5 m^3·tonne^{-1}

Table 3
Summary of organic waste types from various agro-industry sectors and their typical current uses.

Sector	Waste type	Typical current utilisation	Waste availability in Australia[a]	Assumed biogas yield metric[b]	Total biochemical methane potential (m^3_N $CH_4 \cdot annum^{-1}$)[c]
Dairy on-farm	Manure as liquid effluent or as scraped solid/semi-solid	Spread onsite to offset fertiliser use.	475 kilotonnes (kt) $VS \cdot annum^{-1}$	$B_0 = 200\ m^3_N\ CH_4 \cdot t^{-1}\ VS_{added}$	94,951,000
Milk processing	Moderate strength liquid effluent	Liquid effluent irrigated onto agricultural land to offset fertiliser use, or disposed to sewer.	14.96 $GL \cdot annum^{-1}$ liquid effluent	Varies greatly with strength. Assumed 4.5 g $COD \cdot L^{-1}$ (GHD, 2017); COD conversion = 80% (Nadais et al., 2010); i.e. 1.26 $L_N\ CH_4 \cdot L^{-1}$ effluent	18,850,000
	Whey	Whey is disposed to sewer, to piggeries as animal feed, or is irrigated onto farmland	0.78 $GL \cdot annum^{-1}$ whey	Varies greatly with strength. 20 $L_N\ CH_4 \cdot L^{-1}{}_{whey}$	15,640,000
Pork on-farm	Manure as liquid effluent	Effluent treated onsite Spread onsite to offset fertiliser use	187.3 kt $VS \cdot annum^{-1}$ liquid effluent	Assumed $B_0 = 300\ m^3_N\ CH_4 \cdot\ t^{-1}\ VS_{added}$	56,180,000
	Manure as spent bedding	Stockpiled for passive composting Spread onsite/offsite to offset fertiliser use	91 kt $VS \cdot annum^{-1}$ spent straw-based litter	Assumed $B_0 = 200\ m^3_N\ CH_4 \cdot t^{-1}\ VS_{added}$	18,263,000
Beef feedlots	Manure scraped from pens	Composted onsite/offsite for use to offset fertiliser use	273 kt $VS \cdot annum^{-1}$	Assumed $B_0 = 173\ m^3_N\ CH_4 \cdot t^{-1}\ VS_{added}$, for clean and frequently harvested manure	47,240,000
Red meat processing	Moderate to high strength combined liquid effluent	Liquid effluent treated onsite Spread onsite to offset fertiliser use	20.4 $GL \cdot annum^{-1}$ from beef and veal 5.95 $GL \cdot annum^{-1}$ from sheep and mutton	Varies greatly with strength. Assumed 3.7 $m^3_N\ CH_4 \cdot m^{-3}$ effluent	76,380,000 for beef and veal processing 22,280,000 for sheep and mutton processing
	Paunch contents	Composted onsite or offsite for use to offset fertiliser use.	21 kt $VS \cdot annum^{-1}$ for cattle	$B_0 = 242\ m^3_N\ CH_4 \cdot t^{-1}\ VS_{added}$	5,080,000

[a] Estimated in the current work by the method in Section 4.
[b] Refer to Tables 1 and 2 for reported ranges from which the nominal values used in the current work were selected.
[c] Normal (N) gas conditions are 1 atm and 0 °C used throughout this work. The corresponding energy content for methane at these conditions is 39 MJ m^{-3}_N CH_4.

hot standard carcass weight (HSCW) (Ridoutt et al., 2015). Accordingly, national beef and veal production and sheep and mutton production with the above stated annual carcass weights would result in combined liquid effluent amounts given in Table 3. The amount of paunch contents produced from cattle processed across Australia has been estimated by Jensen et al. (2016) at 0.2 $Mt \cdot annum^{-1}$, and assuming a VS content of 10.5% based on characteristics data of Jensen et al. (2016), this gave an equivalent VS amount from the current calculations as given in Table 3.

Total energy demand of the various agro-industry sectors was also estimated in the current work as outlined in Table 4. The results are presented in Fig. 6 together with the primary energy equivalents of the biogas potentials given in Table 3.

The results in Fig. 6 show that milk and meat processing are more energy intensive than upstream on-farm (dairy and piggeries) and intensive feeding (beef feedlots) operations. The latter also appeared to have significant biogas energy potential, with excess energy being potentially available for export (See Section 5.2). Implications of these observations for aggregated AcoD are further discussed in Section 5.3. The form of energy that would be required is important (Table 4). This is because the conversion efficiency of primary energy into electrical energy is typically around 36% as per (Commonwealth of Australia, 2008).

5. Aggregate anaerobic co-digestion opportunities and constraints

Important opportunities and constraints for AD or aggregated AcoD of agro-industry organic waste are discussed in this section.

5.1. Anaerobic co-digestion feedstock benefits and constraints

Whilst transportation costs continue to strongly influence selection of feedstocks for aggregate AcoD facilities (Mata-Alvarez et al., 2014), it is important to formulate feedstock mixtures that support stable and optimum AD performance (Mata-Alvarez et al., 2014). This will logically aim to increase organic loading rate (OLR) (more organic matter can result in more biogas energy produced, up to a certain OLR limit) but can also aim to harness a so-called "priming effect" to stimulate microbial activity (Insam and Markt, 2016), or to dilute chemical inhibitors, provide macro-and micro-nutrient equilibrium, facilitate moisture balance, and provide alkalinity to stabilise digestion pH (Mata-Alvarez et al., 2014). Relevant examples for the current work include:

1. An observed increase in FOG digestion by co-digestion with paunch contents due to dilution of LCFA and/or by provision of lipid-degrading biomass (Astals et al., 2014);
2. Improved digestion of whey by co-digestion with manure, where the whey promotes rapid fermentation and the manure provides nitrogen to minimise inhibition by fermenters (Desai et al., 1994) likely due to the pH buffering effects of ammoniacal nitrogen; and
3. Improved digestion of wheat straw by co-digestion with chicken manure and/or dairy manure (Wang et al., 2012).

Carbon-to-nitrogen ratio (C/N) has long been a popular quantitative metric for formulating co-digestion feedstock mixtures, with pertinent early work originating as far back as 1979 (Hills, 1979). Specifically, carbon-rich substrates provide easily biodegradable organic matter (e.g. glycerol (Astals et al., 2013)) to boost organic loading and to stimulate microbial activity (See this section, above), whilst nitrogen-rich substrates (e.g. manures) provide useful pH buffering at nitrogen concentrations below inhibitory levels (Hagos et al., 2017). This has led to a suggested optimum C/N value for AD between 20 and 30 to promote macro-nutrient balance and digestion stability (Hagos et al., 2017). Notwithstanding that AD is sensitive to several other factors not completely captured by C/N ratio. For example, such factors that can influence AcoD performance such as having too little (Demirel and Scherer, 2011)

Table 4
Methodology summary for estimation of industry energy demand.

Sector	Basis	Energy-demand metric
Dairy on-farm	8.8 GL·annum^{-1} milk produced and processed (including into drinking milk) (Dairy Australia, 2018a)	48 kWh·kL^{-1} milk (173 MJ kL^{-1} milk) (Dairy Australia, 2018b) Mostly electricity for milk cooling (42%), milk harvesting (21%), hot water production (17%), cleaning and effluent systems (9%), stock water supply (9%), shed lighting (4%) and feeding (3%) (Dairy Australia, 2018b).[a]
Milk processing		623–683 MJ kL^{-1} milk processed into non-powder products (Prasad, 2006) Used 658 MJ kL^{-1} milk Energy is typically sourced from natural gas (68%), grid electricity (26%), biomass (2%), coal (3%) and other (1%) (Prasad, 2006).
Pork on-farm	2,972,500 SPUs 90% housed indoors	Estimated average of 40 kWh·SPU^{-1}·annum^{-1} (144 MJ·SPU^{-1}· annum^{-1}) calculated using data given by McGahan et al. (2014) for piggeries with all progeny housed onsite and without a feed mill. Electricity is majority energy type used (75%), followed by diesel (15%), and other (McGahan et al., 2014).
Beef feedlots	1,031,324 head (Table 5.C.8, (Commonwealth of Australia, 2018))	444-1,483 MJ·head^{-1} (Davis and Watts, 2011) Used mid-range value of 965 MJ·head^{-1} Feed management contributes on average 80% of total for those with steam flaking and 45% of total for those without (Davis and Watts, 2011).
Red meat processing	2.4 Mt·annum^{-1} carcass weight beef and veal + 0.7 Mt·annum^{-1} carcass weight sheep and mutton (Meat and Livestock Australia, 2019b).	3,005 MJ t^{-1} HSCW (Ridoutt et al., 2015). About 70% for thermal energy in rendering (Ridoutt et al., 2015). Note, not all Australian RMP facilities have onsite rendering.

[a] Excludes energy for irrigation and fuel consumption for farm operations and transport.

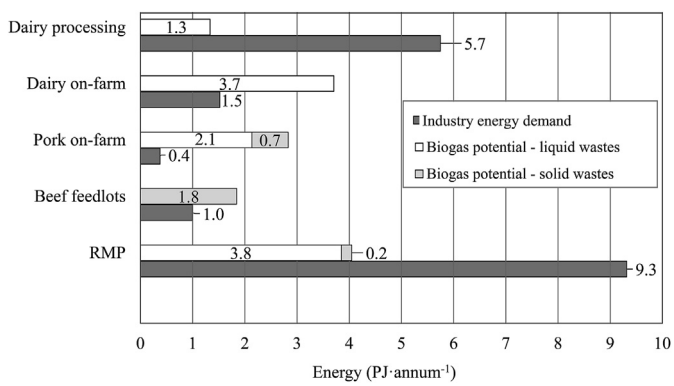

Fig. 6. Energy estimates for Australian agro-industry sectors, including: (dark grey) total sector energy demand; as well total biogas energy potential in liquid (white) and solid (light grey) organic waste from each sector. Liquid waste types include combined liquid effluents from dairy processing, dairy on-farm, pork on-farm and red meat processing, and whey from dairy processing; solid waste include scraped manure from beef feedlots, spent straw-based piggery litter, and cattle paunch from RMP.

or too much trace elements (Romero-Guiza et al., 2016), substrate chemical composition and biodegradability (Hagos et al., 2017), and OLR limits. Consequently, C/N ratio alone may become inadequate to identify preferred co-digestion feedstock mixtures.

Feedstock heterogeneity and inconsistent supply can be significant challenges (Hagos et al., 2017), causing discrepancies between AD performance under controlled lab-scale investigations vs. industrial-scale applications (Hagos et al., 2017). Laboratory batch experiments can characterize biochemical methane potential and degradation rates (Hagos et al., 2017), whilst continuous digester studies are important for industrial applications to consider OLR (Hagos et al., 2017) and identify OLR limits. Unfortunately, such studies are typically more difficult and costly to perform than batch experiments and thus tend to be less prominent. This may be a reason why many biogas plants experience loading problems. Future investigations will be important, especially to understand OLR limits for various digester types, and also considering the impacts of operational temperature and adapted microbial communities. For example, this would be especially important for CAPs becoming increasingly popular in Australian RMP and pork (Section 3). CAPs offer minimal to no ability to control the AD process, are poorly tolerant of complex and high-solids feedstocks (Section 3), and may exhibit temperature-dependent loading limits (Schmidt et al., 2019). Pretreatment and two-stage digestion approaches in general seem promising and could be further explored, especially to provide process intensification (Jensen et al., 2016). The application of pretreatment to high-FOG RMP industry waste has also been reviewed with the view to address operational problems such as pipeline blockages, adhesion to sludge, and inhibition of biological processes (Harris and McCabe, 2015). However, future testing and development are required to identify practical, effective, and economical pretreatment options. For example, energy consumption for thermo-baric pretreatment, specifically to heat up high-moisture RMP fatty waste has been shown to be significant (Harris et al., 2017) and reduces the net energy benefits from recovered biogas. Waste heat sources such as heat recovered from a combined heat and power (CHP) unit, could be considered to improve the economics of thermo-baric pretreatment of fatty RMP waste (Harris et al., 2017).

5.2. Energy benefits and constraints of aggregate anaerobic co-digestion

Biogas installations in Australia have predominantly generated electricity and/or produced heat from biogas in generators or CHP units (McCabe, 2016). Renewable electricity in Australia has been financially encouraged via the Renewable Energy Target (RET) which aimed to achieve 20% of electricity from renewables by 2020 (Edwards et al., 2015), and issues tradeable renewable generation certificates with a historic sale value of ~39 AUD·MWh^{-1} (Edwards et al., 2015). However, because Australia is a net energy exporter (Edwards et al., 2015), renewables and AD have typically been sidelined as fossil fuel-derived business provided relatively cost-effective energy domestically and significant revenue and employment via energy exports (Edwards et al., 2015). Accordingly, in Australia the general demand for renewable electricity, including from biogas, has been comparatively low. Moreover, AD installations have typically been offered relatively low feed-in tariffs (Edwards et al., 2015) (e.g. 0.04–0.08 AUD·kWh^{-1} (AEMO, 2020)). Projects involving on-site "behind-the-meter" have typically demonstrate better financial viability (Carlu et al., 2019). This is because a higher tariff is typically paid for purchased energy, also including the offset of supply and distribution charges.

CHP units using biogas to produce both heat and electricity can diversify energy types available for use and thereby improve overall use efficiency. For example, a commonly assumed electrical conversion efficiency for internal combustion engine generators is 36% (Commonwealth of Australia, 2019), but a CHP unit can make at least as much useful heat energy available as electrical energy. This is relevant for agro-industries reviewed in the current work, wherein both electricity and heat are needed (See Section 4). There is a need for future work to explore cost-effective absorption chilling options to utilise biogas directly and provide chilling at a relevant scale, such as for example in dairy on-farm for milk cooling (Birchall et al., 2008). This could save significant capital costs by negating the need for an intermittent electricity generation step, because electricity would then not be required to provide the cooling. Most of the agro-industries reviewed in the current work have need for cooling (See Section 4).

5.3. Future opportunities and challenges for aggregate anaerobic co-digestion

Aggregated AcoD of agro-industrial organic waste may provide significant benefits and opportunities for municipal wastewater treatment plants (WWTPs). This is because;

1. Unlike agro-industries for which waste management is usually an activity aside from core business, WWTPs have traditionally been designed to provide end-of-pipe solutions for liquid waste, specifically domestic wastewater (Nghiem et al., 2017);
2. Operations at WWTPs typically include AD to digest biological sludge produced from sewage and treatment processes (Nghiem et al., 2017). Consequently, the majority of existing AD facilities in Australia are actually at WWTPs (McCabe, 2016), treating sewage sludge;
3. Thickened sewage sludge consists of about 95% of water, so that OLRs of digesters at WWTPs have been typically hydraulically limited to below 1 kg $VS \cdot m^{-3} \cdot d^{-1}$ (Nghiem et al., 2017). Also, WWTPs are commonly designed for future capacity increases, meaning that significant spare digestion capacity could be available for AcoD (Nghiem et al., 2017); and
4. Wastewater treatment processes are energy intensive, so that AcoD of imported organic matter-rich agro-industrial waste could help WWTPs progress towards future energy-neutrality (Nghiem et al., 2017).

AcoD at WWTPs seems particularly logical, considering that several Australian agro-industrial facilities already discharge liquid waste into sewers (e.g. dairy processing and RMP, Section 3), which often ends up at WWTPs. Industrial discharges into sewers can carry odour potential, which influences effective odour abatement options at the downstream WWTPs (Fisher et al., 2018). Instead, the option of by-passing sewers could be considered, transporting suitable feedstocks rich in organic-matter directly to anaerobic digesters located at WWTPs to maximise energy recovery and reduce negative impacts.

Opportunities for AcoD are also available within agro-industry sectors. For example, some Australian piggeries import food processing waste and past-use-by-date food products (with due consideration of biosecurity) to supplement purchased grain diets (Australian Pork Newspaper, 2014). This can provide more cost-effective pig diets, and where waste is received but unable to be used in pig feed, these can boost biogas production from an onsite CAP (See Section 3). This could be especially beneficial where such a piggery is located close to a waste-source industry.

Food processing facilities are frequently located on the fringes of urban centers (e.g. RMP facilities (Australian Meat Processors Corporation, 2010) and milk processing (Liu and Haynes, 2011)) or in rural areas close to the milk supply in the case of milk processing producing longer shelf life products (Liu and Haynes, 2011). Estimates of biogas energy availability and energy demand showed that off-farm processing (RMP and dairy processing) is notably more energy intensive than on-farm and intensive feeding production, which in-turn have significant biogas energy availability (See Section 4). Accordingly, it could be attractive to locate centralised AcoD facilities in the vicinity of or at processing facilities.

As highlighted in Section 3, various waste feedstocks pose unique opportunities and challenges to AcoD performance and influences the suitability of various digestion technologies. In addition to materials handling considerations (e.g. pre-heating of FOG-rich substrates to prevent solidification at lower temperatures (Nghiem et al., 2017)), a high nutrient content of AcoD substrates is also important as it may require additional treatment capacity for centrate management (Nghiem et al., 2017). This would affect the amount and cost of digestate post-handling to prevent adverse environmental impacts. Relevant regulatory frameworks for AcoD are still in early stages of development and as a result digestate may be subject to stringent and conservative regulations for transport and handling (Nghiem et al., 2017). AD releases/mobilises nutrients previously bound to organic matter. This can enable subsequent recovery into formulated, transportable and saleable fertiliser products (Mehta et al., 2015). The formulation of balanced fertilizer products will be important to make transport and beneficial use of nutrients cost-effective. This is an area for further research and development to support sustainable closed-loop concepts (Antille et al., 2018). Future work should explore the fate of pathogens in AcoD to identify cost-effective means to reliably produce safe digestates from agro-industrial organic waste. For example, whilst AD at ambient and mesophilic temperatures may not be completely effective for pathogen inactivation (Jiang et al., 2020), the integration of solid-state digestion approaches with VFA/ammonia accumulation could be considered as a promising option for pathogen inactivation (Jiang et al., 2020).

Landfill levies have been introduced in Australia at the State level to facilitate the diversion of organic waste away from landfill, and this can provide gate-fee revenue to an AcoD facility (Edwards et al., 2015). Also, Australian biogas projects have considerably benefited from emissions reduction incentives (e.g. ERF) and the RET (See Section 5.2). Related project grant funding and attractive renewable energy finance options have also been available (Carlu et al., 2019). Further policy and funding support would similarly encourage biogas production into the future, potentially also for conversion into renewable natural gas (Carlu et al., 2019). However, future work should aim to better understand the impacts of relevant policies, and in collaboration with industry and governments develop and implement policies that could support a sustainable environmental business from aggregated AcoD (Carlu et al., 2019).

6. Conclusions

Australia's red meat, dairy, and pork industries produce significant quantities of organic waste during on-farm production, intensive feeding and processing of animals (estimated at ~79 $GL \cdot annum^{-1}$ liquid waste plus ~2 $megatonnes \cdot annum^{-1}$ solid waste). Understanding key information gaps on waste composition and quantities in these industries is a fundamental step towards fully realizing the opportunities for biogas energy from agro-industrial organic waste streams via anaerobic digestion. This review provided a critical cross-industry evaluation of available information on such waste streams and revealed varying extents of information but good biogas energy potential (estimate ~13.8 PJ. $annum^{-1}$).

Important data gaps identified by the review and which influence anaerobic digestion and aggregate anaerobic co-digestion opportunities include: (a) methane potential of Australian dairy effluent; (b) Australian data sets on solid and liquid waste streams from dairy processing; (c) data on the proportion of Australian pig deep litter systems on straw vs. other bedding types; (d) and data on organic loading limits of different digester types for agro-industrial waste.

Manure is an important organic waste type from pork, beef feedlots and dairy on-farm, and is produced either as a liquid or solid waste, which influences anaerobic processing options. Australian dairy production on-farm is predominantly pasture-based, with only a portion of daily manure output recoverable, and this influences the feasibility of anaerobic digestion. Infrequent and coarse manure harvesting practices at Australian beef feedlots make the collected manure less amenable to biogas production. It may be possible to change part of the manure management practices at a beef feedlot to produce "cleaner" manure with a higher methane yield.

In terms of technology, covered anaerobic ponds are becoming increasingly popular for liquid organic waste in Australia, because of relatively low cost and simplicity, but offer minimal ability to control the digestion performance and have a poor tolerance of complex organic waste such as solid organic waste from red meat processing. Moderate-to-high strength liquid waste produced by agro-industry sectors may be amenable to onsite high-rate anaerobic treatment. Pretreatment should be further explored for process intensification to facilitate anaerobic processing of complex high-strength and high-solids organic waste, particularly if covered ponds are used, and also considering the cost and parasitic energy load of such pretreatment.

Potential benefits for agro-industries from behind-the-meter use of biogas energy are significant (estimate energy demand at ~18 PJ. annum^{-1}) and has the ability to significantly decrease production costs and increase industry profitability. This could also enable the diversion of organic waste away from landfill and further reduce environmental footprint such as via greenhouse gas mitigation and renewable energy generation. Municipal wastewater treatment plants could notably benefit from importing and aggregated co-digestion of agro-industrial organic waste to progress towards energy neutrality. Whilst a biogas project should ideally be feasible based on energy value alone, on-going and future policy incentives would greatly encourage biogas projects and enable greenhouse gas abatement via anaerobic co-digestion in Australia.

CRediT author statement

Stephan Tait: Conceptualization; Investigation; Formal analysis; Writing- Original draft preparation.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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References

Acil Allen Consulting, 2017. Economic Contribution Report: Pork Industry in Australia 2015-16. Report Prepared for Australian Pork Limited. Available at: http://australianpork.com.au/wp-content/uploads/2018/01/ACILAllen_PorkIndustryEconomicImpactReport_Final_17Nov2017_corrected.pdf. (Accessed 22 April 2020).

Aemo, 2020. NEM data dashboard. Available at: https://aemo.com.au/en/energy-systems/electricity/national-electricity-market-nem/data-nem/data-dashboard-nem. (Accessed 18 May 2020).

Allinson, G., Halliwell, D.J., Stokes, J., 2007. Closing the loop on a large, multi-disciplinary dairy waste project. Aust. J. Dairy Technol. 62 (3), 135–153.

Angelidaki, I., Sanders, W., 2004. Assessment of the anaerobic biodegradability of macropollutants. Rev. Environ. Sci. Biotechnol. 3, 117–129.

Antille, D.L., McCabe, B.K., Van Der Spek, W., Baillie, C.P., 2018. An investigation into the fertilizer potential of slaughterhouse cattle paunch. Trans. ASABE (Am. Soc. Agric. Biol. Eng.) 61 (1), 87–101.

Antonopoulou, G., Stamatelatou, K., Venetsaneas, N., Kornaros, M., Lyberatos, G., 2008. Biohydrogen and methane production from cheese whey in a two-stage anaerobic process. Ind. Eng. Chem. Res. 47 (15), 5227–5233.

Arcadis, 2019. National Food Waste Baseline - Australia. Final assessment report prepared for the Department of Agriculture, Water and the Environment. Available at: https://www.environment.gov.au/protection/waste-resource-recovery/food-waste. (Accessed 9 April 2020).

Astals, S., Batstone, D.J., Mata-Alvarez, J., Jensen, P.D., 2014. Identification of synergistic impacts during anaerobic co-digestion of organic wastes. Bioresour. Technol. 169, 421–427.

Astals, S., Musenze, R.S., Bai, X., Tannock, S., Tait, S., Pratt, S., Jensen, P.D., 2015. Anaerobic co-digestion of pig manure and algae: impact of intracellular algal products recovery on co-digestion performance. Bioresour. Technol. 181, 97–104.

Astals, S., Nolla-Ardevol, V., Mata-Alvarez, J., 2013. Thermophilic co-digestion of pig manure and crude glycerol: process performance and digestate stability. J. Biotechnol. 166 (3), 97–104.

Australian Clean Energy Regulator, 2020. Emissions reduction fund project register. Available at: http://www.cleanenergyregulator.gov.au/ERF/project-and-contracts-registers/project-register. (Accessed 23 April 2020).

Australian Meat Processors Corporation, 2010. Introduction to the Red Meat Processing Industry.

Australian Pork Newspaper, 2014. Kia-Ora Piggery – Finalist in the Victorian Premier's Sustainability Awards 2014, October edition. page 9. Cleveland, Australia, 4163. Available at: http://porknews.com.au/documents/pasteditions/APN1014.pdf. (Accessed 12 July 2020).

Bai, M., Flesch, T., Trouve, R., Coates, T., Butterly, C., Bhatta, B., Hill, J., Chen, D.L., 2020. Gas emissions during cattle manure composting and stockpiling. J. Environ. Qual. 49 (1), 228–235.

Banks, C.J., Wang, Z., 2004. Treatment of meat wastes. In: Banks, C.J., Wang, Z. (Eds.), Waste Treatment in the Food Processing Industry. CRC Press, pp. 11–38. https://doi.org/10.1201/9781420037128-4.

Baskaran, K., Palmowski, L.M., Watson, B.M., IWA Programme Committee, 2003. Wastewater Reuse and Treatment Options for the Dairy Industry, 3rd World Water Congress: Efficient Water Supply and Water Reuse, pp. 85–91.

Batstone, D.J., Jensen, P.D., 2011. Anaerobic Processes. Treatise on Water Science, vol. 4. Water-Quality Engineering, pp. 615–639.

Birchall, S., 2010. Biogas Production by Covered Lagoons — Performance Data from Bears Lagoon Piggery.

Birchall, S., Dillon, C., Wrigley, R., 2008. Effluent and Manure Management Database for the Australian Dairy Industry. Dairy Australia.

Britz, T.J., van Schalkwyk, C., Hung, Y.-T., 2004. Waste treatment in the food processing industry. In: Wang, L.K., Hung, Y.-T., Lo, H.H., Yapijakis, C. (Eds.), Waste Treatment in the Food Processing Industry. CRC Press, pp. 11–38. https://doi.org/10.1201/9781420037128-4.

Carlu, E., Truong, T., Kundevski, M., 2019. Biogas Opportunities for Australia. MARCH 2019. A Report Prepared by ENEA Consulting, for Bioenergy Australia, the Australian Renewable Energy Agency, Energy Networks Australia, IEA Bioenergy and Clean Energy Finance Corporation. Available at: https://www.bioenergyaustralia.org.au/news/biogas-opportunities-for-australia-report-release/. (Accessed 3 June 2020).

Carvalho, F., Prazeres, A.R., Rivas, J., 2013. Cheese whey wastewater: characterization and treatment. Sci. Total Environ. 445, 385–396.

Chen, G.Q., Talebi, S., Gras, S.L., Weeks, M., Kentish, S.E., 2018. A review of salty waste stream management in the Australian dairy industry. J. Environ. Manag. 224, 406–413.

Christie, K.M., Rawnsley, R.P., Phelps, C., Eckard, R.J., 2018. Revised greenhouse-gas emissions from Australian dairy farms following application of updated methodology. Anim. Prod. Sci. 58 (5), 937–942.

Commonwealth of Australia, 2008. National Greenhouse and Energy Reporting (Measurement) Determination 2008. Section 2.38. Available at: https://www.legislation.gov.au/Details/F2017C00508/Controls/. (Accessed 23 May 2020).
Commonwealth of Australia, 2018. National inventory Report 2018 volume 1. Available at: https://www.industry.gov.au/sites/default/files/2020-05/nga-national-inventory-report-2018-volume-1.pdf. Last accessed 11/12/2020.
Commonwealth of Australia, 2019. Carbon Credits (Carbon Farming Initiative—Animal Effluent Management) Methodology Determination 2019. Available at: https://www.legislation.gov.au/Details/F2020L00005. (Accessed 23 April 2020).
Craggs, R., Park, J., Heubeck, S., 2008. Methane emissions from anaerobic ponds on a piggery and a dairy farm in New Zealand. Aust. J. Exp. Agric. 48 (1–2), 142–146.
Dairy Australia, 2017. A guide for investment and the Australian dairy industry. Available at: https://www.dairyaustralia.com.au/about-dairy-australia/about-the-industry/investment-and-the-australian-dairy-industry. (Accessed 2 April 2020).
Dairy Australia, 2018a. Dairy at a Glance 2018-19. Available at: https://www.dairyaustralia.com.au/industry/farm-facts/dairy-at-a-glance. (Accessed 2 April 2020).
Dairy Australia, 2018b. Saving energy on dairy farms. Available at: https://www.dairyingfortomorrow.com.au/wp-content/uploads/1437-Saving-energy-on-dairy-farms-Booklet-2018_FA_DIGITAL_20181210.pdf. (Accessed 23 May 2020).
Dairy Australia, 2019a. Cows and farms. Available at: https://www.dairyaustralia.com.au/industry/farm-facts/cows-and-farms. (Accessed 2 April 2020).
Dairy Australia, 2019b. In focus 2019. The Australian dairy industry. Available at: https://www.dairyaustralia.com.au/industry/farm-facts/in-focus. (Accessed 13 May 2020).
Dairy Australia, 2021. The Australian dairy industry. Available at: https://www.dairyaustralia.com.au/about/the-australian-dairy-industry#.YFL87NwRWUk. (Accessed 18 March 2021).
Davis, R.J., Watts, P.J., 2011. Quantifying the Water and Energy Usage of Individual Activities within Australian Feedlots. Part B Report: Energy Usage at Australian Feedlots. Report Prepared for Meat and Livestock Australia Project B.FLOT.0339. Available at: https://www.mla.com.au/research-and-development/search-rd-reports/final-report-details/Environment-On-Farm/Quantifying-the-Water-and-Energy-Usage-of-Individual-Activities-within-Australian-Feedlots/264. (Accessed 23 May 2020).
Davis, R.J., Watts, P.J., McGahan, E.J., 2012. Quantification of Feedlot Manure Output for BEEF-BAL Model Upgrade, RIRDC Project No. PRJ–004377, Rural Industries Research and Development Corporation. Canberra, ACT. Available at: https://www.agrifutures.com.au/product/quantification-of-feedlot-manure-output-for-beef-bal-model-upgrade/. (Accessed 29 April 2020).
Demirel, B., Scherer, P., 2011. Trace element requirements of agricultural biogas digesters during biological conversion of renewable biomass to methane. Biomass Bioenergy 35 (3), 992–998.
Department of Environment and Energy, 2019. Australian energy update - Australian energy statistics september 2019. Available at: https://www.energy.gov.au/publications/australian-energy-update-2019. (Accessed 18 May 2020).
Desai, M., Patel, V., Madamwar, D., 1994. Effect of temperature and retention time on biomethanation of cheese whey-poultry waste-cattle dung. Environ. Pollut. 83 (3), 311–315.
Durham, R.J., Hourigan, J.A., 2009. Waste management and co-product recovery in dairy processing. In: Waldron, K. (Ed.), Handbook of Waste Management and Co-product Recovery in Food Processing, vol. 1. Woodhead Publishing Limited and CRC Press LLC, pp. 332–387.
Edwards, J., Othman, M., Burn, S., 2015. A review of policy drivers and barriers for the use of anaerobic digestion in Europe, the United States and Australia. Renew. Sustain. Energy Rev. 52, 815–828.
Erguder, T.H., Tezel, U., Guven, E., Demirer, G.N., 2001. Anaerobic biotransformation and methane generation potential of cheese whey in batch and UASB reactors. Waste Manag. 21 (7), 643–650.
Fernandez-Gutierrez, D., Veillette, M., Giroir-Fendler, A., Ramirez, A., Faucheux, N., Heitz, M., 2017. Biovalorization of saccharides derived from industrial wastes such as whey: a review. Rev. Environ. Sci. Biotechnol. 16 (1), 147–174.
Fisher, R., Shammay, A., Alvarez-Gaitan, J.P., Stuetz, R.M., 2018. Sewer catchment effects on wastewater and biosolids odour management. Water Sci. Technol. 77 (9), 2348–2354.
Fredheim, L., 2018. Oil and Grease Value Assessment Tool. A Report Prepared for the Australian Meat Processor Corporation. Project 2018-1023.
Fyfe, J., Hagare, D., Sivakumar, M., 2016. Dairy shed effluent treatment and recycling: effluent characteristics and performance. J. Environ. Manag. 180, 133–146.
GHD, 2017. Anaerobic Digestion as a Treatment and Energy Recovery Technology for Dairy Processing Waste Streams. A Report Prepared for Dairy Australia. Available at: https://manufacturing.dairyaustralia.com.au. (Accessed 13 May 2020).
Gopalan, P., Jensen, P.D., Batstone, D.J., 2013a. Anaerobic digestion of swine effluent: impact of production stages. Biomass Bioenergy 48, 121–129.
Gopalan, P., Jensen, P.D., Batstone, D.J., 2013b. Biochemical methane potential of beef feedlot manure: impact of manure age and storage. J. Environ. Qual. 42 (4), 1205–1212.
Graham, P., 2020. Lot Feeding Brief. Results for the December Quarter 2019 Feedlot Survey. Meat & Livestock Australia. **Available at:**. https://www.mla.com.au/globalassets/mla-corporate/prices–markets/documents/trends–analysis/lot-feeding-brief/mla_lot-feeding-brief_may-2019.pdf. (Accessed 27 April 2020).
Hagos, K., Zong, J.P., Li, D.X., Liu, C., Lu, X.H., 2017. Anaerobic co-digestion process for biogas production: progress, challenges and perspectives. Renew. Sustain. Energy Rev. 76, 1485–1496.
Harris, P.W., McCabe, B.K., 2015. Review of pre-treatments used in anaerobic digestion and their potential application in high-fat cattle slaughterhouse wastewater. Appl. Energy 155, 560–575.
Harris, P.W., Schmidt, T., McCabe, B.K., 2017. Evaluation of chemical, thermobaric and thermochemical pre-treatment on anaerobic digestion of high-fat cattle slaughterhouse waste. Bioresour. Technol. 244, 605–610.
Hassan, A.N., Nelson, B.K., 2012. Invited review: anaerobic fermentation of dairy food wastewater. J. Dairy Sci. 95 (11), 6188–6203.
Hauser, J., 2017. Analysis of South Australia Whey Processing Options. Dairy Australia Project. Horticulture Australia. Australian Horticulture Statistics Handbook 2016/17.
Heubeck, S., Craggs, R.J., 2010. Biogas recovery from a temperate climate covered anaerobic pond. Water Sci. Technol. 61 (4), 1019–1026.
Hills, D.J., 1979. Effects of carbon-nitrogen ratio on anaerobic-digestion of dairy manure. Agric. Wastes 1 (4), 267–278.
Hjorth, M., Christensen, K.V., Christensen, M.L., Sommer, S.G., 2010. Solid-liquid separation of animal slurry in theory and practice. A review. Agron. Sustain. Dev. 30 (1), 153–180.
IEA Bioenergy Task 37, 2020. IEA Bioenergy Task 37 – Country Reports Summaries 2019. Available at: https://task37.ieabioenergy.com/country-reports.html. (Accessed 12 July 2020).
Ince, O., 1998. Potential energy production from anaerobic digestion of dairy wastewater. J Environ Sci and Health Part a-Toxic/Hazardous Substances & Environmental Engineering 33 (6), 1219–1228.
Insam, H., Markt, R., 2016. Comment on "Synergistic co-digestion of solid-organic-waste and municipal-sewage-sludge: 1 plus 1 equals more than 2 in terms of biogas production and solids reduction. Water Res. 87, 416–423. Water Res 95, 392-393.
Jacobs, J.L., Ward, G.N., Kearney, G., 2008. The effect of dairy effluent on dry matter yields, nutritive characteristics, and mineral content of summer-active regrowth forage crops in southern Australia. Aust. J. Agric. Res. 59 (6), 578–588.
Janke, L., McCabe, B.K., Harris, P., Hill, A., Lee, S., Weinrich, S., Marchuk, S., Baillie, C., 2019. Ensiling fermentation reveals pre-treatment effects for anaerobic digestion of sugarcane biomass: an assessment of ensiling additives on methane potential. Bioresour. Technol. 279, 398–403.
Jensen, P., Sullivan, T., Carney, C., Batstone, D., 2014. Analysis of the potential to recover energy and nutrient resources from cattle slaughterhouses in Australia by employing anaerobic digestion. Appl. Energy 136, 23–31.
Jensen, P.D., Mehta, C.M., Carney, C., Batstone, D.J., 2016. Recovery of energy and nutrient resources from cattle paunch waste using temperature phased anaerobic digestion. Waste Manag. 51, 72–80.
Jensen, P.D., Yap, S.D., Boyle-Gotla, A., Janoschka, J., Carney, C., Pidou, M., Batstone, D.J., 2015. Anaerobic membrane bioreactors enable high rate treatment of slaughterhouse wastewater. Biochem. Eng. J. 97, 132–141.
Jiang, Y., Xie, S.H., Dennehy, C., Lawlor, P.G., Hu, Z.H., Wu, G.X., Zhan, X.M., Gardiner, E., 2020. Inactivation of pathogens in anaerobic digestion systems for converting biowastes to bioenergy: a review. Renew. Sustain. Energy Rev. 120.
Johns, M.R., 2012. Waste Water Management in the Australian Red Meat Processing Industry. Guidelines for the Day-To-Day Management and Operation of Upstream Waste Water Treatment, Anaerobic Ponds, Methane Gas Capture Systems, and Aerobic Ponds. National Meat Industry Training Advisory Council (MINTRAC).
Kruger, I., Taylor, G., Roese, G., Payne, H., 2006. PRIMEFACT 68: Deep-Litter Housing for Pigs.
Kushwaha, J.P., Srivastava, V.C., Mall, I.D., 2011. An overview of various technologies for the treatment of dairy wastewaters. Crit. Rev. Food Sci. Nutr. 51 (5), 442–452.
Labatut, R.A., Angenent, L.T., Scott, N.R., 2011. Biochemical methane potential and biodegradability of complex organic substrates. Bioresour. Technol. 102 (3), 2255–2264.
Li, Y.B., Park, S.Y., Zhu, J.Y., 2011. Solid-state anaerobic digestion for methane production from organic waste. Renew. Sustain. Energy Rev. 15 (1), 821–826.
Liu, Y.Y., Haynes, R.J., 2011. Origin, nature and treatment of effluents from dairy and meat processing factories and the effects of their irrigation on the quality of agricultural soils. Crit. Rev. Environ. Sci. Technol. 41 (17), 1531–1599.
Longhurst, R.D., Roberts, A.H., O'Connor, M.B., 2000. Farm dairy effluent: a review of published data on chemical and physical characteristics in New Zealand. NZJAR (N. Z. J. Agric. Res.) 43, 7–14.
Mainardis, M., Flaibani, S., Trigatti, M., Goi, D., 2019. Techno-economic feasibility of anaerobic digestion of cheese whey in small Italian dairies and effect of ultrasound pre-treatment on methane yield. J. Environ. Manag. 246, 557–563.
Mason, I.G., 1997. Performance of a facultative waste stabilization pond treating dairy shed wastewater. Trans. ASAE (Am. Soc. Agric. Eng.) 40 (1), 211–218.
Mata-Alvarez, J., Dosta, J., Romero-Güiza, M.S., Fonoll, X., Peces, M., Astals, S., 2014. A critical review on anaerobic co-digestion achievements between 2010 and 2013. Renew. Sustain. Energy Rev. 36, 412–427, 0.
McCabe, B.K., 2016. IEA Bioenergy Task 37. Country Report Australia. Availabe at: http://task37.ieabioenergy.com/country-reports.html?file=files/daten-redaktion/download/publications/country-reports/2016/Task%2037%20Country%20Report%20Australia%202016.pdf. (Accessed 17 May 2020).
McCabe, B.K., Hamawand, I., Harris, P., Baillie, C., Yusaf, T., 2014. A case study for biogas generation from covered anaerobic ponds treating abattoir wastewater: investigation of pond performance and potential biogas production. Appl.

Energy 114, 798–808.
McCabe, B.K., Harris, P., Antille, D.L., Schmidt, T., Lee, S., Hill, A., Baillie, C., 2020. Toward profitable and sustainable bioresource management in the Australian red meat processing industry: a critical review and illustrative case study. Crit. Rev. Environ. Sci. Technol. 50 (22), 2415–2439. Published online. https://doi-org.ezproxy.library.uq.edu.au/10.1080/10643389.2020.1712310.
McCabe, B.K., McMeniman, J., 2017. MLA Tips & Tools - feasibility of using feedlot manure for biogas production. Prepared for Meat and Livestock Australia (MLA). Available at: https://www.mla.com.au/globalassets/mla-corporate/research-and-development/documents/presentations/tipstools_feasibility_052017.pdf. (Accessed November 2018).
McCabe, B.K., Tait, S., Marchuk, S., Lee, S., Antille, D.L., Eberhard, J., Baillie, C., 2019. Identification and Analysis of Organic Waste Produced from Australian Intensive Livestock Industries: Opportunities for Aggregation and Treatment via Co-digestion, 2019 ASABE Annual International Meeting. ASABE, Boston, Massachusetts. https://doi.org/10.13031/aim.201900170. July 7–10. Published online. https://doi-org.ezproxy.library.uq.edu.au/10.1080/10643389.2020.1712310.
McGahan, E.J., Phillips, F.A., Wiedemann, S.G., Naylor, T.A., Warren, B., Murphy, C.M., Griffith, D.W.T., Desservettaz, M., 2016. Methane, nitrous oxide and ammonia emissions from an Australian piggery with short and long hydraulic retention-time effluent storage. Anim. Prod. Sci. 56 (9), 1376–1389.
McGahan, E.J., Warren, B.R., Davis, R.J., 2014. Establishing energy usage on Australian piggeries to enable implementation of energy reduction strategies. In: Final Report Prepared for Australian Pork Limited Project No. 2012/2407. Available online. Last accessed. (Accessed 23 May 2020).
Meat & Livestock Australia, 2012. National Beef Cattle Feedlot Environmental Code of Practice, second ed. xii. Available at: www.mla.com.au Last accessed 13/12/2020.
Meat & Livestock Australia, 2019a. Fast Facts: Australia's beef industry. In: Australia, M.a.L. (Ed.), Meat and Livestock Australia. Available at: https://www.mla.com.au/globalassets/mla-corporate/prices–markets/documents/trends–analysis/fast-facts–maps/mla-beef-fast-facts-2019.pdf. (Accessed 27 April 2020).
Meat & Livestock Australia, 2019b. State of the Industry Report. The Australian Red Meat and Livestock Industry. Available at: https://www.mla.com.au/globalassets/mla-corporate/prices–markets/documents/trends–analysis/soti-report/mla-state-of-industry-report-2019.pdf. (Accessed 24 April 2020).
Meat & Livestock Australia, 2020. State of the Industry Report. The Australian Red Meat and Livestock Industry. Available at: https://www.mla.com.au/globalassets/mla-corporate/prices–markets/documents/trends–analysis/soti-report/mla-state-of-industry-report-2020.pdf. Last accessed 13/12/2020.
Mehta, C., Tucker, R., Poad, G., Davis, R., McGahan, E., Galloway, J., O'Keefe, M., Trigger, R., Batstone, D., 2016. Nutrients in Australian agro-industrial residues: production, characteristics and mapping. Australas. J. Environ. Manag. 23 (2), 206–222.
Mehta, C.M., Khunjar, W.O., Nguyen, V., Tait, S., Batstone, D.J., 2015. Technologies to recover nutrients from waste streams: a critical review. Crit. Rev. Environ. Sci. Technol. 45 (4), 385–427.
Meng, L.Y., Maruo, K., Xie, L., Riya, S., Terada, A., Hosomi, M., 2019. Comparison of leachate percolation and immersion using different inoculation strategies in thermophilic solid-state anaerobic digestion of pig urine and rice straw. Bioresour. Technol. 277, 216–220.
Mittal, G.S., 2006. Treatment of wastewater from abattoirs before land application - a review. Bioresour. Technol. 97 (9), 1119–1135.
Moller, H.B., Sommer, S.G., Ahring, B.K., 2002. Separation efficiency and particle size distribution in relation to manure type and storage conditions. Bioresour. Technol. 85 (2), 189–196.
Mussoline, W., Esposito, G., Lens, P., Garuti, G., Giordano, A., 2014. Electrical energy production and operational strategies from a farm-scale anaerobic batch reactor loaded with rice straw and piggery wastewater. Renew. Energy 62, 399–406.
Nadais, M.H.G.A.G., Capela, M.I.A.P.F., Arroja, L.M.G.A., Hung, Y.-T., 2010. Anaerobic treatment of milk processing wastewater. In: Wang, L.K., Tay, J.-H., Tay, S.T.L., Hung, Y.-T. (Eds.), Handbook of Environmental Engineering, vol. 11. Environmental Bioengineering, Humana Press, Totowa, NJ. https://doi.org/10.1007/978-1-60327-031-1.
Nennich, T.D., Harrison, J.H., VanWieringen, L.M., Meyer, D., Heinrichs, A.J., Weiss, W.P., St-Pierre, N.R., Kincaid, R.L., Davidson, D.L., Block, E., 2005. Prediction of manure and nutrient excretion from dairy cattle. J. Dairy Sci. 88 (10), 3721–3733.
Nghiem, L.D., Koch, K., Bolzonella, D., Drewes, J.E., 2017. Full scale co-digestion of wastewater sludge and food waste: bottlenecks and possibilities. Renew. Sustain. Energy Rev. 72, 354–362.
O'Hara, I., Robins, K., Forde, G., Henry, B., Jensen, P., Speight, R., McNicholl, D., 2016. Research, Development and Adoption Strategy for Environmental Innovation within the Australian Red Meat Supply Chain. Meat and Livestock Australia Limited, Australia. Available from: https://eprints.qut.edu.au/123505/1/123505.pdf. (Accessed 2 April 2020).
Phelps, C., Heubeck, S., Craggs, R.. Is biogas technology right for Australian dairy farms? N.D. https://www.dairyaustralia.com.au. (Accessed 8 April 2020).
Prasad, P., 2006. Australian Dairy Manufacturing Industry - State of the Environment Report. A Dairy Australia Report on Behalf of the Dairy Manufacturers Sustainability Council. Available at: https://www.ecoefficiencygroup.com.au/wp-content/uploads/2017/11/2005-Australian-Dairy-Manufacturing-Industry-State-of-the-Environment-Report.pdf. (Accessed 8 April 2020).
Prasad, P., Pagan, R., Kauter, M., Price, N., 2005. Eco-efficiency for the Dairy Processing Industry. Dairy Australia, Southbank, Victoria.
Pratt, C., Redding, M., Hill, J., Mudge, S.R., Westermann, M., Paungfoo-Lonhienne, C., Schmidt, S., 2014. Assessing refrigerating and freezing effects on the biological/chemical composition of two livestock manures. Agric. Ecosyst. Environ. 197, 288–292.
Prazeres, A.R., Carvalho, F., Rivas, J., 2012. Cheese whey management: a review. J. Environ. Manag. 110, 48–68.
Rico, C., Rico, J.L., Garcia, H., Garcia, P.A., 2012. Solid-liquid separation of dairy manure: distribution of components and methane production. Biomass Bioenergy 39, 370–377.
Ridoutt, B., Sanguansri, P., Alexander, D., 2015. Environmental Performance Review: Red Meat Processing Sector 2015. Prepared for the Australian Meat Processor Corporation. Australian Meat Processor Corporation.
Romero-Guiza, M.S., Vila, J.J., Mata-Alvarez, J., Chimenos, J.M., Astals, S., 2016. The role of additives on anaerobic digestion: a review. Renew. Sustain. Energy Rev. 58, 1486–1499.
Schmidt, T., Harris, P., Lee, S., McCabe, B.K., 2019. Investigating the impact of seasonal temperature variation on biogas production from covered anaerobic lagoons treating slaughterhouse wastewater using lab scale studies. J Environ Chem Eng 7 (3).
Sibilio, S., Rosato, A., Ciampi, G., Scorpio, M., Akisawa, A., 2017. Building-integrated trigeneration system: energy, environmental and economic dynamic performance assessment for Italian residential applications. Renew. Sustain. Energy Rev. 68, 920–933.
Skerman, A., Collman, G., Pott, L., 2011. Biogas Production by Covered Lagoons QNPH Piggery Grantham Queensland.
Skerman, A.G., Heubeck, S., Batstone, D.J., Tait, S., 2018. On-farm trials of practical options for hydrogen sulphide removal from piggery biogas. Process Saf. Environ. Protect. 117, 675–683.
Skerman, A.G., Tait, S., 2018. Bioenergy support program - DAF transition. Final report prepared for Pork CRC Project 4C-116.
Skerman, A.G., Willis, S., Tait, S., Yap, S.D., 2016. Effect of Feed Wastage on Piggery Manure Characteristics and Methane Potential. Final report prepared for APL Project 2015/010. Available at: https://australianpork.infoservices.com.au/downloads/2015-010. (Accessed 6 March 2020).
Tait, S., Tamis, J., Edgerton, B., Batstone, D.J., 2009. Anaerobic digestion of spent bedding from deep litter piggery housing. Bioresour. Technol. 100 (7), 2210–2218.
Tucker, R.W., 2015. APL Project 2012/1028, Piggery Manure and Effluent Management and Reuse Guidelines.
Tucker, R.W., 2018. National Environmental Guidelines for Indoor Piggeries, third ed. Australian Pork Limited, Kingston, ACT, Australia. APL Project 2015-2221.
Tucker, R.W., McDonald, S., O'Keefe, M., Craddock, T., Galloway, J., 2015. Beef Cattle Feedlots: Waste Management and Utilisation. Meat & Livestock Australia, North Sydney, NSW, Australia.
Vivekanand, V., Mulat, D.G., Eijsink, V.G.H., Horn, S.J., 2018. Synergistic effects of anaerobic co-digestion of whey, manure and fish ensilage. Bioresour. Technol. 249, 35–41.
Wang, X.J., Yang, G.H., Feng, Y.Z., Ren, G.X., Han, X.H., 2012. Optimizing feeding composition and carbon-nitrogen ratios for improved methane yield during anaerobic co-digestion of dairy, chicken manure and wheat straw. Bioresour. Technol. 120, 78–83.
Watson, P., Watson, D., 2015. Down to earth research. Sustainability Framework NRM Survey. Dairy Australia.
Watts, P., McCabe, B., 2015. Feasibility of Using Feedlot Manure for Biogas Production. Meat and Livestock Australia.
White, T., John, M., Butler, B., 2013. Methane recovery and use at a meat processing facility - King Island. Final Report Prepared Fro RIRDC Project No. PRJ-005673.
Wiedemann, S.G., McGahan, E.J., Murphy, C.M., 2016. Environmental impacts and resource use from Australian pork production assessed using life-cycle assessment. 1. Greenhouse gas emissions. Anim. Prod. Sci. 56 (9), 1418–1431.
Wilkinson, K.G., Brooks, R.B., Balmer, C., Halliwell, D., Palmowski, L., Issa, J.G., Jeyaseelan, S., Meehan, B., Baskaran, K., 2007. A survey of waste management practices in Victorian dairy factories. Aust. J. Dairy Technol. 62 (3), 154–158.
Yap, S.D., 2017. Anaerobic Treatment of Solid Manure Residues. A Thesis Submitted for the Degree of Doctor of Philosophy at. The University of Queensland, School of Chemical Engineering and Advanced Water Management Centre.
Yap, S.D., Astals, S., Jensen, P.D., Batstone, D.J., Tait, S., 2016. Pilot-scale testing of a leachbed for anaerobic digestion of livestock residues on-farm. Waste Manag. 50, 300–308.
Yap, S.D., Astals, S., Jensen, P.D., Batstone, D.J., Tait, S., 2017. Indigenous microbial capability in solid manure residues to start-up solid-phase anaerobic digesters. Waste Manag. 64, 79–87.