Life cycle assessment of renewable methane production technologies

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Abstract

This study aims to understand environmental impacts of renewable methane production from seven different technologies including wind- and solar-powered electrolysis and methanation with direct air-captured CO_2, anaerobic digestion of sewage sludge, pig and dairy manure, and food waste, as well as landfill gas upgrade. Life cycle impact assessment using ReCiPe 2016 method and scenario analysis were performed. Sewage sludge system exhibited the highest global warming impact due to electricity consumption during digestate dewatering by centrifuge, whereas anaerobic digestion of pig manure in a covered lagoon with dewatering in Geotube® demonstrated the lowest impact. When centrifuge and Geotube® were compared for the same production system, Geotube® had 78% lower global warming impact than dewatering with centrifuge. In general, less energy and resource requirements of major processes led to lower impact together with environmental credits assigned to avoided products, such as NPK fertiliser and conventional manure management, as well as air-captured CO_2. Significantly higher human toxicity and ecotoxicity impacts of wind- and solar-powered renewable methane systems were detected due to material and energy consumption during construction of the solar panels and wind farms. Increasing efficiency of wind and solar electricity appeared to be more sensitive than increasing efficiency of electrolyser, while transport distance of the landfill gas had significant effect on the global warming impacts. The potential for producing renewable methane from waste materials is dependent on various supports, such as carbon credits, construction of infrastructure and amendment of national gas standard to accelerate the production.

Highlights

- Environmental impacts of seven renewable methane production systems are compared
- Sewage sludge-based renewable methane exhibits the highest global warming impact
- Low energy consumption during the production lead to the low environmental impacts
- Environmental credits provided to avoided products partially compensate the impact
- Efficiency of wind and solar electricity is one of the key parameters

Keywords: Life cycle assessment, renewable methane production, renewable natural gas, synthetic natural gas, waste to energy, power to gas, power to methane, biogas, waste to energy, environmental impact assessment

Word count: 9229

Abbreviations

AD	Anaerobic digestion	**LFG**	Landfill gas
AED	Anaerobic digestion in engineered digester	**NPK**	Nitrogen, phosphorous and potassium
AK	Alkaline	**PEM**	Proton exchange membrane
CAL	Covered anaerobic lagoon	**PtG**	Power-to-gas
CHP	Combined heat and power	**PtM**	Power-to-methane
C/N ratio	Carbon-nitrogen ratio	**RNG**	Renewable natural gas
DAC	Direct air capture	**SNG**	Synthetic natural gas
ERF	Emissions Reduction Fund	**SPU**	Standard pig unit
HDPE	High-density polyethylene	**TS**	Total solid
LCA	Life cycle assessment	**VS**	Versatile solid
LCI	Life cycle inventory	**WWTP**	Wastewater treatment plant

1. Introduction

The Increasing amount of waste due to economic development and population increase highlights the importance of waste management, preferably reducing the amount of waste [1]. Although 16-67% of municipal solid waste in the OECD countries is recycled with increasing trend of the recycling, 100% of waste recycling is unrealistic [2]. During the period of 2020-2021, a total of 75.8 million tonnes (Mt) of waste was generated in Australia with 48.1 Mt of organic waste, of which 55% was recycled and reused, while 15.3% was landfilled [3,4].

Biogas can be produced by decomposition of organic waste, such as animal manure, food waste and sewage sludge, by anaerobic digestion (AD), which is recommended by IPCC for waste management not only to convert the waste materials into energy products but also to reduce emissions from the waste management [1,5,6]. Despite the potential of the organic waste as a source of biogas production, only minor fraction of the waste was recovered in Australia, with 1.6% of 4.7 Mt food organics anaerobically digested, and 37.6% of total landfill gas captured and energy-recovered in 2020-2021 [3,4]. There are currently 247 biogas production facilities in Australia, mostly located in south-eastern coastal areas (Fig. 1a), with more than half based

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on landfill gas (130 facilities), 52 facilities are wastewater treatment plants (WWTP), whereas 20 facilities are pig manure-based biogas production (out of 22 agricultural biogas facilities, Fig. 1b). Given that total 5,515 kt of animal waste and 201 kt of sewage sludge were produced, while 35% of total biogas was flared in 2020-2021, there is a great potential of increasing energy value from the biogas production in Australia [7,8], Also, Australian livestock sector has been identified as an area that requires emission reduction, particularly emissions from manure management. AD of animal manure can be a solution to reduce the emissions by altering current manure management system and simultaneously producing renewable energy [9-11].

Driven by net zero emission target by 2050 and the planned phase-out of coal power stations which are expected to be halved by 2030, renewable electricity generation in Australia has almost doubled during the last five years accounting for 29% of the total electricity generation in 2021 [12-14]. Increasing trend of the renewable electricity generation is mainly driven by solar PV and wind with share of 40.1% and 34.5% in total renewables in 2021, respectively [15]. However, relatively low production capacity, and intermittent nature of solar and wind energy sources have emphasised the need for increasing share of other types of renewable energy, such as biogas and biomethane [16]. Hydrogen is also considered as a promising storage medium by converting the wind and solar energy to a liquid or gaseous form [17].

Renewable hydrogen can be produced by splitting water in electrolyser powered by wind or solar electricity. The produced hydrogen can later be used to produce renewable methane by Sabatier reaction (Eq. 1) where hydrogen and carbon dioxide react with the aid of catalyst [18]. The carbon dioxide can be captured from fossil-based power stations or from renewable sources including biogas plant and the atmosphere [19,20].

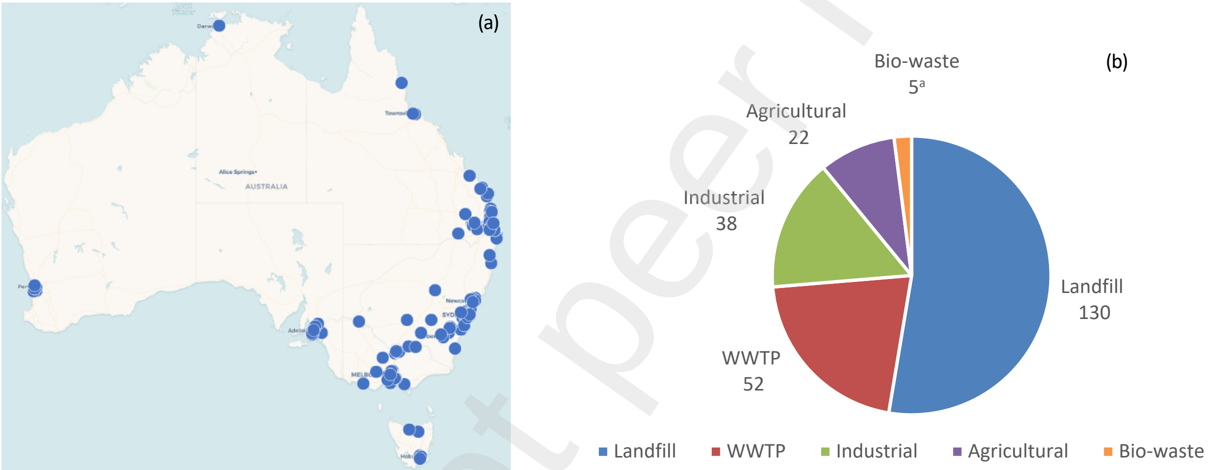

Fig. 1. Biogas production in Australia; (a) Location of 247 biogas facilities; (b) Number of facilities according to biogas source

a. Numbers below the biogas source indicate the number of biogas facilities

* Source: 7, 21.

A study by Gerloff [19] compared different sources of carbon dioxide and electricity for renewable methane production from electrolysis with methanation processes which was often referred to as Power-to-Methane (PtM) or Power-to-Gas (PtG) [22]. In this study, carbon dioxide capture from the air, cement and biogas plant, and three types of electrolysers, alkaline, polymer electrolysis membrane and solid oxide electrolysis cell were analysed with grid and 100% renewable electricity. When the same grid electricity was used, the PtM process with biogas plant-captured carbon dioxide had 1.7% higher CO_2 equivalent emissions than the case with air-captured carbon dioxide. This study also pointed out that emissions from the PtM technology was only lower than conventional natural gas when renewable electricity was used.

$$4H_2 + CO_2 \rightarrow CH_4 + 2H_2O, \quad \Delta H = -165\ kJ/mol \qquad \text{Eq. (1)}$$

Renewable methane is also termed as renewable natural gas (RNG) or synthetic natural gas (SNG) because chemical composition of the gas is comparable to natural gas [19,23], where methane is the major component (usually more than 95%) with minor impurities, such as hydrogen sulphide and moisture [24-46]. Unlike hydrogen, renewable methane has a number of advantages. Due to the similarity of the chemical composition, renewable methane can substitute for natural gas by transmission via existing natural gas pipelines, indicating reduction in natural gas use and less requirement for constructing new infrastructure for the transmission [13.27]. In addition, there is no threshold for renewable methane to be injected into the natural gas grid, whereas depending on gas standard of each country, about 2-15% of hydrogen can be blended with the natural gas [28,29]. Major composition of biogas comprises of 54-73% of CH_4, 30-39% CO_2, 1-10% moisture and less than 1% of H_2S and N_2 [24,30,31], while required CH_4 content in the biogas differs according to biogas combustion technologies, with CH_4 content between 50-60% for combustion in engines and higher than 55% in micro turbines [32,33]. However, for the biogas to be injected into the grid, impurities should be removed by biogas upgrading, for instance H_2S content less than 5.7 mg/m^3 and O_2 content below 0.2 mol % are required by the Australian Specifications for General Purpose of Natural Gas (AS 4564) [25]. A number of biogas upgrading technologies are available with different technological requirements for each technology (Table 1). In terms of process energy, amine scrubbing is known to have the lowest electricity requirement, but its heat and chemical demand can make the technology less favourable in some cases. Upgrading by water scrubbing is the most used, while membrane technology is recently increasing in use due to its cost-efficiency and simplicity in operation with almost 60% share of these two technologies in the global renewable methane production [1,34,35].

The role of biogas and renewable methane has been recognised in many reports as a solution to achieve emissions reduction target with great potential for production increase. A report by the International Energy Agency (IEA) [1] estimates that 570 million tonnes of oil equivalent (Mtoe) of biogas and 730 Mtoe of biomethane can be produced globally using renewable sources, which can potentially supply 20% of global gas demand, but only 35 Mtoe (sum of biogas and biomethane production) was produced in 2018. Several countries have already realised the value of biomethane, mostly in European countries. Germany is reported to have the highest number of biogas upgrading facilities with 232 operating in 2019 and more than 90% of the facilities grid-connected. France also has more than 90% of grid-connected facilities out of the total of 131 facilities in 2020 [36]. Australia does not have grid-connected biomethane facilities yet, although recent report by Australian Energy Market Operator (AEMO) projected greater share of biomethane under the 1.5°C Green Energy Exports scenario driven by rapid reduction in fossil gas production from 2023 and onwards [13]. Australian Renewable Energy Agency (ARENA) [37] claimed that about 105 PJ of renewable gas, such as renewable methane, could be integrated within the existing natural gas pipelines, which would account for 23% of the total pipeline gas market.

Table 1 Biogas upgrading technologies with characteristics

Literature average	Water scrubbing	Chemical scrubbing	Membrane	Pressure swing adsorption
Methane content after upgrading (%)	95-98	98-99	95-97	96-98
Electricity demand (kWh/m^3 biogas)	0.2-0.5	0.1-0.3	0.2-0.35	0.2-0.43
Heat demand (MJ/m^3 biogas)	n.r.[a]	1.5-1.83	n.r.	n.r.
Water demand (kg/m^3 biogas)	40-200[b]	0.55	n.r.	n.r.
Chemical demand (kg/m^3 biogas)	n.r.	0.05	n.r.	n.r.
Operation pressure (bar)	4-10	0.1-3	5-20[c]	2-10
Operation temperature (°C)	40	100-180	Ambient-40	70
Methane loss (%)	0.5-5	0.1-4	0.5-2	1.5-2.5
Complexity of operation	Medium	Complex	Simple	Medium
Operation cost	Low	Medium to high	Low	High
Investment cost	Low	Medium	Low to medium	High

Values used in this study	Water scrubbing	Chemical scrubbing (amine)	Membrane	Pressure swing adsorption
Electricity demand (kWh/m^3 biogas)	**0.26**[d]	**0.16**	**0.27**	Not analysed
Heat demand (MJ/m^3 biogas)	n.r.	**1.6**	n.r.	
Water demand (kg/m^3 biogas)	**45**	**0.55**	n.r.	
Amine demand (kg/m^3 biogas)	n.r.	**0.05**	n.r.	

a. n.r.: not required.
b. Water demand varies depending on water circulation. 40-80 kg of water is required per m^3 of biogas when water circulation is applied, whereas 100-200 kg is required when is not applied.
c. High operation pressure is found in high-pressure membrane systems.
d. Values in **bold** are used in this study.
* Source: 26, 34, 35, 38- 42, 43-46.

Waste-produced biogas has also been discussed in academic literature. One of the recent studies performed regionalised life cycle assessment (LCA) for food waste treated by AD and incineration, with possible uses of the produced biogas in Singapore. AD of food waste presented better environmental performances than incineration, while distance for the waste collection was identified as one of the most influential factors for the assessment [47]. Similar findings were presented in the study by Slorach et al. [48] who estimated about 36% of emission savings from AD of food waste instead of incineration could be achieved, while Dastjerdi et al. [49] reported 270 kg CO_2 eq of avoided emissions per year from AD of food waste instead of landfilling. To seek a better option for biogas yield, various mixtures of feedstock have been tested. For example, Fusi et al. [50] compared the environmental impacts of biogas produced by mono- and co-digestion of pig and cow slurry with residues of tomato and maize. The impacts were presented according to electricity (per MWh) generated by combustion of the produced biogas in combined heat and power (CHP) unit that was sold to the grid, excluding heat generated from the CHP using part of the produced biogas. The study concluded that mono-digestion of cow slurry with solid content of 8.5% had the greatest environmental benefit followed by co-digestion of pig slurry with maize silage. Despite the detailed inventory data presented in this study, production potential of the biogas was not fully investigated because the leftover heat (heat produced from the CHP unit minus heat used for AD process) was considered as waste, hence the impact per unit of electricity did not fully reflect energy content of the biogas. Also, energy value can vary depending on chemical composition of the biogas and electrical efficiency of CHP, but when these factors are not defined in the study, the impact assessment result per unit of electricity can be misinterpreted. Likewise, many previous studies apply different functional units (e.g., GJ of biogas, MWh of electricity) with different low and high heating values, but when variable energy content ranging between 16-28 MJ per m^3 of biogas and 9.8-11 kWh per m^3 of renewable methane with the methane density of 0.67 kg/m^3 (at 15 °C and 1 bar) and 3.38 kg/m^3 (at 15 °C and 5 bar) are considered, impact per unit of energy or volume can be difficult to understand unless conversion factors under specific conditions are clearly presented [30,50,51].

Limited system boundaries have been presented in power-to-gas studies where production of hydrogen was the final stage of the boundary. Also, biogas studies were often performed as part of waste management, with process flows and environmental impacts expressed in per unit of waste without system expansion up to the biogas upgrading for renewable methane production, which was only discussed in a limited number of studies [52-54]. LCA studies are particularly lacking in renewable methane production from waste materials, instead studies have mostly focused on technological aspects of biogas production and upgrading, usually with a single feedstock, or reviewed multiple production systems without providing environmental burdens and benefits for a particular production system [35,44,55]. Also, a range of factors that affect the methane yield, such as physical and chemical characteristics of feedstock, level of pre-treatment and operation conditions, are not always available among the LCA studies. This study attempts to fill the gaps in the literature by performing LCA of various renewable methane production technologies with seven different feedstocks (resources) based on extensive inventory data developed by aggregating multiple data sources, including literature, commercial database, and data from manufactures and operators. To the authors' best knowledge, no LCA work about renewable methane production has been performed that encompasses a range of production technologies, resources and waste materials with a focus on maximum production potential of the renewable methane without integrating flare and CHP unit for a partial use of the produced methane. Through the assessment, each production system will be understood based on the environmental impacts with identifying key parameters that have most influence on the impacts. Significance of each key parameter is later confirmed by scenario analysis, while discrepancies of some determinant factors, as well as challenges and opportunities, are further discussed based on case studies and statistical data.

2. Methods

This study is structured based on ISO 14040 standards which are; 1. Goal and scope definition, 2. Life cycle inventory, 3. Life cycle assessment and 4. Interpretation [56]. Detailed processes of each component are provided in the following sections.

2.1. Goal and scope definition

2.1.1. Goal and functional unit

The aim of this study is to evaluate the environmental impacts of seven different renewable methane production technologies by system modelling using various data sources. The study objectives also include identifying key parameters that affect the impacts with suggesting optimal combination of the parameters. In this study, 1 kg of renewable methane (expressed as kg CH_4) leaving the methanation and biogas upgrading facility was selected as a functional unit to understand the environmental impacts of a unit of methane and for convenience of unit conversion.

2.1.1. System boundary

System boundary of both wind- and solar PV-based production systems begins with provision of raw materials for manufacture of wind turbines and solar panels, whereas supply of sewage sludge after primary treatment in wastewater treatment plant (WWTP) is included as the first stage of the sewage sludge system (Fig. 2). The difference in the system boundary is due to the assumptions made for each production system. Wind and solar power station was assumed to be integrated into the renewable methane production by methanation of hydrogen and carbon dioxide, with associated production facilities assumed to be constructed for the purpose of methane production and therefore included in the system boundary, while primary treatment of sewage sludge was considered as part of conventional wastewater treatment, thus construction of the WWTP with the primary treatment was excluded. Anaerobic digestor and lagoon for digestion of the selected feedstocks, and biogas upgrading facility were included for both construction and operation. Some process flows and avoided processes were modelled following the real Australian cases and Ecoinvent database v3.8 [57] when literature data conflicted, while final use of the produced methane, decommissioning of plant, and recycling or disposal of plant materials were excluded in this study. Further details of the modelling process of each production system are described in Section 2.2.1 – 2.2.6.

2.1.3. System expansion

All the modelled systems include at least one avoided or recycled product along with renewable methane as the major system product. For this multi-functional system, system expansion approach (also referred as avoided burden or substitution approach) was applied in order to assign environmental credits to the product that could either be recycled within the modelled system or could substitute for commercial product that could otherwise be produced in a conventional way [58,59].

2.2. Life cycle inventory (LCI) and system modelling

Inventory analysis was performed in two steps. First, both primary (data from a technical report of manufacturer and plant operator) and secondary data (data from literature and LCA databases) were aggregated from various sources (Table S1 in Supplementary material). Use of primary data was prioritised for foreground processes (for classification of foreground and background processes, see Table S1), and proxy data compatible with the included processes were used when relevant data were not available from the primary sources. According to LCA guidelines, proxy data can be used when the primary and secondary data are not available [59,60]. Minor modification was performed for some datasets in order to better reflect the reality, which included average efficiency of wind and solar electricity generation adjusted for Australian wind and solar farms by excluding the lowest 10% efficiencies reported in the literature, while biogas yield from the lagoon system and total solid content (TS) in the manure were adjusted based on reports by the Australian livestock industry [23,32,61,62]. The finalised inventory data were later validated

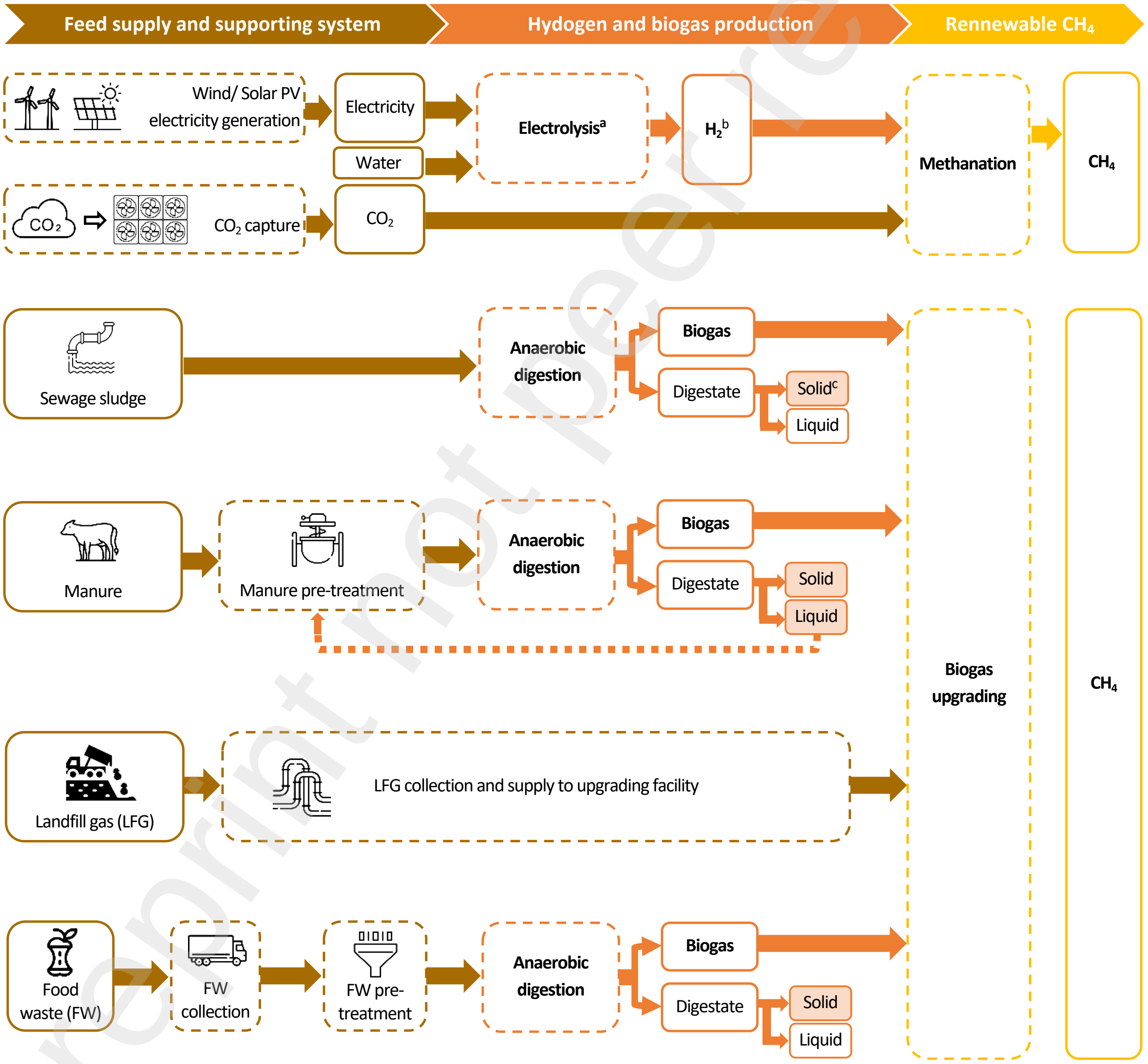

Fig. 2. System boundary of the renewable methane technologies assessed in this study

a. Boxes with a dashed line represent processes.
b. Boxes with a solid line represent products.
c. Colour-filled boxes represent avoided products.

by relevant process flows in Ecoinvent database v3.8 and industrial data from real Australian cases. The Ecoinvent data were used for most background processes, except centrifuge unit, which was averaged from Agribalyse database v3.0.1 and literature data (for detailed data sources, see Table S1 in Supplementary material). For the major source of process energy, 2020 Australian grid electricity and Australian natural gas were used, except for wind and solar systems. The grid mix consisted of 42.2% black coal, 20.8% natural gas, 12.7% brown coal and 1.7% diesel, comprising 77.4% of the total grid mix, while the remaining 22.6% was generated from renewables including 7.9% solar PV, 7.7% wind, 5.7% hydro, 0.8% bagasse and wood, and 0.5% from biogas [63]. Anaerobic digestion in engineered digester (AED) was modelled as single-stage digester with minor heat supply for maintaining the treatment temperature for mesophilic condition (temperature between 35-38 °C), whereas lagoon system was assumed to operate in ambient temperature without heat supply. For biogas upgrading, membrane separation technology was used as a reference case for all the biogas-based systems, which was later compared with upgrading by water scrubbing and amine scrubbing technologies as alternative scenarios. Sulphur content in the biogas was removed by activated carbon during upgrading, while chiller with condensate collection trap was used for moisture removal from the biogas [24,51,64-66]. CO_2 separated by the membrane upgrading was assumed as biogenic CO_2 with zero emission factor. Since there was no commercial facility for renewable methane production from biogas upgrading and methanation, hypothetical processes were developed based on available data.

2.2.1. System modelling of wind-based methane production

Detailed material and energy flows associated with manufacture of wind turbine as well as construction and operation of wind power station were mostly modelled according to the real Australian wind farm (Musselroe Wind Farm, TAS, Australia) and wind turbine manufacturer used in the wind farm (Vestas 3MW wind turbine) [67-69], which was later validated with the relevant process flows in Ecoinvent v3.8. Wind to electricity conversion efficiency was calculated based on literature and low wind case of the wind farm because the low wind case was within the range of reported value in the literature [67,70-74]. High wind case of the wind farm was later modelled as a scenario. The electrolysis processes were modelled based on the Hydrogen Park South Australia (HyP SA) that was installed with proton exchange membrane (PEM) electrolyser. Together with the PEM, alkaline (AK) electrolyser was also modelled as a reference case for both wind- and solar-based systems by maintaining other parameters the same (W-pem.ref and W-ae.ref for wind, and S-pem.ref and S-ae.ref for solar). The major inventory data with references and process flows of the modelled systems are presented Table 2, and Table S1 and S2 in Supplementary material. Since construction and operation of wind and solar power station, as well as process energy requirement of power-to-gas systems have been identified as key parameters [19,75], high efficiency scenarios were developed for both wind and solar systems (W1 and S1 in Table 5, respectively). In order to understand the influence of electricity source on the overall impact, the reference wind system was modelled with 2020 Australian grid electricity (W2 in Table 5), while electrolysers with high and low efficiencies were applied to the reference case by creating additional scenarios (W3 and W4 in Table 5).

Finally, renewable methane was produced by Sabatier reaction of hydrogen from wind- and solar-powered electrolysis, and carbon dioxide captured from the atmosphere (direct air capture, DAC). Technical processes and inventories for DAC unit were modelled based on Climeworks, the world's largest CO_2 capture plant operator [76,77]. Temperature–vacuum swing adsorption with amine-based sorbent were applied as the CO_2 capture technology. The data from Climeworks were obtained through personal communication that included both current and future production capacities, of which, 4 kt capacity (current) plant was used in this analysis [77]. Construction of methanation reactor and its operation processes were modelled based on literature with H_2 to CO_2 ratio for producing 1 kg of renewable methane calculated as 0.5 kg H_2 : 2.75 kg CO_2 with the methanation efficiency of 80% [18-20,30,34,40,76,78-85]. Thermo-chemical methanation process was applied in this study which consumed nickel-based catalyst in a catalytic adsorption reactor. Waste heat produced from the exothermic methanation process (165 kJ/mol of heat, see Eq. 1) was assumed to be recycled as process energy of DAC unit, thus heat requirement that could otherwise be provided by natural gas was considered as avoided product. Lifetime of the wind power plant, electrolyser and methanation reactor was assumed to be 20 years.

2.2.2. System modelling of solar PV-based methane production

Darlington Point Solar Farm (NSW, Australia) was selected as a reference case with the inventories for its construction and operation used in this study. The farm is installed with multi-crystalline PV panels and constructed as open-ground mounting with single-axis tracking system. The farm consists of 826,850 PV panels producing about 685,000 MWh/year with 30 years lifetime of the panels [102-106]. Although the major system components were modelled based on the solar farm, full LCI was constructed by averaging literature data, industrial reports and Ecoinvent v3.8 (see Table 2) [54,88-101]. Conversion efficiency of solar to electricity was averaged to 16.5% (reference case) and higher efficiency of 19.5% was used for scenario analysis [89-92,96,101-105]. The following electrolysis, CO_2 capture and methanation processes remained the same as the wind-based system.

Table 2 Major inventory for reference case of the modelled production system

	System code	Wind/ solar electricity efficiency (%)	PEM and AK electrolyser efficiency (%)	Electricity use for electrolysis (kWh/kg H_2)	Water use for electrolysis (kg/H_2)	Electricity use for CO_2 capture (kWh/kg CO_2)	Sorbent use for CO_2 capture (g/kg CO_2)	H_2:CO_2 ratio for methanation (kg/kg CH_4)	Electricity use for methanation (kWh/kg CH_4)
Wind	**W-pem.ref** [a]	30	61.8	55.4	11.5	0.47	7.5	0.5:2.75	0.45
	W-ae.ref		64	54					
Solar	**S-pem.ref**	16.5	61.8	55.4					
	S-ae.ref		64	54					

		Input feedstock (kg/m^3 biogas)	Output (after digestion, kg/m^3 biogas)	Mixing water (kg/m^3 biogas)	Electricity use for centrifuge (kWh/m^3 CH_4)	TS (% of input feedstock)	VS (% of TS)	N:P:K (% of TS)	CH_4 content in biogas (%)
Sewage sludge	**SEW-cen.ref**	60	52.2	n.r.[b]	4.4.	6	87.5	3.76:1.38:0.4	65
Pig manure	**P-lagoon.ref**	44.2	36.4	5.8	n.r.	9.2	84.8	5.0:1.5:1.8	66
	P-aed.ref	38.4	31.6	n.r.	2.80				
Dairy manure	**D-lagoon.ref**	44.2	36.4	6.2	n.r.	9.3	80.8	4.52:0.94:2.2	63.4
	D-aed.ref	38	31.3	n.r.	2.88				
Landfill gas	**LF-mem.ref**	n.r.	n.r.	n.r.	n.r.	n.r.	n.r.	n.r.	63
Food waste	**FW-cen.ref**	19.1	16.6	4.9	1.46	27	90	2.9:0.5:0.65	62.5

a. For included process flows of each reference system (e.g., W-pem.ref), see Table S2 in Supplementary material.

b. n.r.: not required.

* Source: wind (54, 67, 70-74, 86, 87), solar (54, 88-105), electrolysis (18, 19, 86, 96, 97, 106-114), CO_2 capture (18-20, 76, 77, 81, 85, 115, 116), methanation (18-20, 30, 40, 76, 78, 79, 80, 82, 85), sewage sludge (30, 41, 51, 117-125), pig manure (23, 31, 32, 62, 64, 126-137), dairy manure (5, 23, 32, 40, 50, 61, 62, 118, 126, 128, 131-133, 135, 138-142), landfill gas (27, 42, 43, 143-146), food waste (47, 48, 125, 147-154).

2.2.3. System modelling of methane production from sewage sludge

Malabar Biomethane Project (NSW, Australia) is the first renewable methane project in Australia that plans to update the existing WWTP with additional installation of biogas upgrading unit by membrane separation to produce renewable methane [51]. This facility was referenced for determining major processes, while detailed inventory data were obtained from Ecoinvent v3.8 ('Sewage treatment by anaerobic digestion' process) and literature [30,41,51,117-125]. Current practice at Malabar WWTP includes combustion of biogas in CHP to produce electricity. In this study, the use of the biogas was modified to include biogas upgrading for the purpose of production of renewable methane only. Following AED of the sludge, the digestate was dewatered using two different techniques, Geotube® and centrifuge. Geotube® is a geo-textile, made of woven polypropylene monofilament yarns that is permeable to liquid while facilitating the solid fraction in the digestate to remain within the tube so that it can be removed after dewatering (also termed as solid and liquid separation) [23,137]. In contrast to centrifuge, the separation by Geotube® utilises physical differences between the solid and liquid, thus process energy is not required. Total solid (TS) content of up to 95% and 65% can be recovered after separation by Geotube® and centrifuge, respectively [23,133,137].

Nutrient contents (N, P, and K) in the separated solid fraction were assumed to replace the same amount of commercial N, P and K fertilisers (the amount was calculated using N:P:K contents, 3.76:1.38:0.4 % of TS in the sewage sludge [125], see Table 2), which were later modelled to be applied to adjacent field within 20 km radius. Environmental credits were applied to the substitution of NPK fertilisers, while emissions from the field application process were included as environmental burden. The separated liquid fraction was assumed to be sent to local WWTP through sewer pipelines where the liquid fraction was treated with other wastewater. Since 82% of Australian WWTP provides up to tertiary level of treatment [4], the amount of input sludge used to produce biogas was credited as an avoided wastewater treatment.

The average TS content of 6% of the sludge was used with corresponding amount of input sludge (60 kg of sludge for producing 1 m^3 of biogas) as the reference sewage system that included AED, digestate dewatering by centrifuge and membrane biogas upgrading (SEW-cen.ref in Table 2), while 4% of TS, lower range of literature data with 90 kg of sludge input was modelled as an alternative scenario with the rest of the system components remaining the same as the reference system [41,120-122,125]. Another scenario with dewatering of digestate by Geotube® was developed to confirm influence of the dewatering technique, while maintaining the rest of the system identical to the reference.

2.2.4. System modelling of methane production from animal manure

The methane production from animal manure systems considered in this work are based on biogas production by digestion of pig and dairy cattle manure in a covered anaerobic lagoon (CAL), which represent the most commonly applied system in Australian livestock industry with very few recent installations of engineered anaerobic digestor [127,169]. Accordingly, two reference system models were developed, with the one as manure digestion in CAL with digestate dewatering in Geotube® (P-lagoon.ref for pig and D-lagoon.ref dairy in Table 2), and the other as the digestion in engineered anaerobic digester with dewatering by centrifuge (P-aed.ref and D-aed.ref) for both pig and dairy manure. Reports by Australian livestock organisations (e.g., Australian Pork, Dairy Australia) [9,23,24,32,61,62,66,130,135,155-160] were mostly used for the system modelling of CAL system, while AED system was modelled based on Ecoinvent and literature data (see Table 2) [5,31,40,50,64,118,126-129,131-134,136,138-142]. Pig and dairy manures are usually flushed with water in Australia, thus the manure is collected in a slurry-like form with lower solid content than manure, which represents dry-scrapped animal manure. In this study, the term manure refers to animal manure in a slurry form. Average TS of 9.2% and 9.3% of pig and dairy manure was used, respectively, which were later mixed with water to make the TS to 8% before it was supplied to the CAL [23,40,50,129,131,132,133,136,138,139,142]. 8% of TS is the threshold for manure being able to be pumped, which is particularly important for CAL without any mechanical aids (e.g., stirring) [23]. According to Evans et al. [161], feedstock with solid contents between 10-25% can be used in engineered digester. The average TS contents in the two manures were within this range, thus the modelled AED system did not involve manure mixing with water. Mixing water for the CAL system was assumed to be recycled from the separated liquid fraction after dewatering of the digestate. The amount of mixing water was calculated as 5.8 kg and 6.2 kg for pig and dairy manure, respectively (Table 2).

Part of the solid content is not degraded during the digestion process, thus it accumulates over time at the bottom of the digester which should be removed periodically to prevent decrease in process efficiency caused by reduction of active treatment volume of digester [23]. Pipelines for the accumulated sludge removal were assumed to be installed at the bottom of digester with desludging pump and access point for the removal work. Installation of the pipelines and construction of lagoon were modelled based on the design guidelines by Australian Pork [62] and Meat & Livestock Australia [61]. Volume of the accumulated sludge (that should be removed) was calculated using sludge accumulation rate of 0.00137 m^3/kg TS [62] with loading rate of 0.6 kg VS (versatile solid)/m^3/day and VS content 84.8% and 80.8% of TS for pig and dairy, respectively (for TS and VS contents, see Table 2). Two types of materials were considered for lining and cover of the lagoon, one with high-density polyethylene (HDPE) lining and cover, and the other one with concrete-constructed CAL with HDPE cover, with both CALs equipped with additional pipes for biogas and stormwater collection and electronic control unit for monitoring biogas production and sludge accumulation (for data sources, see Table S1 in Supplementary material). Other system components, such as digestate separation and substitution of fertiliser, remained the same as the other reference systems to understand the influence of the selected parameters only, except for the additional credit assigned to the recycled liquid fraction as a mixing water and the avoided manure treatment in a conventional effluent pond. Avoided emissions were calculated using methane conversion factor (Cf_{CH4}) of 0.75 for the effluent pond with Eq. 2, while emission of nitrous oxide from the pond was reported as negligible, hence it was excluded from the credit [156].

$$E_{\text{CH4}}\,(\text{kg}) = VS * Ef_{\text{CH4}} * Cf_{\text{CH4}} * D_{\text{CH4}} \qquad \text{Eq. (2)}$$

Where E_{CH4} = avoided methane emissions (kg CH_4)
VS = volatile solid content of selected feedstock, estimated using percentage of VS in Table 2
Ef_{CH4} = methane emission factor, 0.45 m^3 CH_4/ kg VS
Cf_{CH4} = methane conversion factor, 0.75
D_{CH4} = methane density, 0.678 kg/m^3

2.2.5. System modelling of methane production from landfill gas

Given that composition of landfill gas (LFG) is similar to biogas with CH_4 content of 63% (from Woodlawn Bioreactor Project, NSW Australia) [162] and the LFG is assumed to be collected from existing landfills installed with LFG collection and leachate treatment systems, the system boundary of the LFG system includes LFG collection and provision to biogas upgrading facility (Fig 2). The system was modelled with construction of LFG collection and conveyance pipelines assumed to have been constructed within 1 km radius as a reference system (LF-mem.ref), which was later increased to 8 km and 20 km. Also, biogas leak during LFG collection and transport was reported as being a major parameter, with 1% and 2% leaks considered as reference and scenario cases, respectively [27,40,42,43,143-146,162]. Considering almost half of the current Australian LFG is flared [163], while the current study considers the whole fraction of the captured LFG is used for production of renewable methane, avoided flaring [42] of the LFG was provided with environmental credit. Biogas upgrading by membrane (reference system) was later compared with alternative scenarios of upgrading by water scrubbing and amine scrubbing [30,34,38-46,121,142].

2.2.6. System modelling of methane production from food waste

The reference system of food waste was modelled based on EarthPower (NSW, Australia), Australia's first food waste-to-energy facility, and literature data, except for the TS content [47,48,125,147-154]. Food waste was assumed to be collected by lorries within 20 km radius based on waste collection points of EarthPower. Upon arrival at the AD facility, non- degradable materials, such as plastics and metals (less than 5% of total food waste), were removed by hydro-pulper [154]. Environmental credits provided to the avoided chemical fertiliser and post-treatment of separated liquid fraction were applied the same as in the other systems for consistency of modelling, which were also practiced in the EarthPower facility. Considering that 83% of food organics were landfilled in 2020-2021 [4], avoided emissions from landfill of food waste were also credited.

Since there were significant differences between TS content from literature [47-49,125,148,149,152,153] and EarthPower [147,154] with 27% and 12%, respectively, literature average was used for the reference system (FW-cen.ref), while 12% TS was modelled as alternative scenario (FW1 in Table 5). The 12% TS was diluted with water to achieve 5% TS for digestion at the EarthPower facility which was modelled accordingly [154], while the reference case was mixed with water to TS 20% which was the highest range of solid content that could be digested in mesophilic conditions [164]. The required amount of food waste for producing 1 m^3 of biogas was calculated using the TS contents and biogas yield of 0.29 m^3/kg VS. Both the reference system and scenario were modelled with AED and centrifuge (For major inventories, see Table 2).

2.3. Life cycle assessment

LCA process involves allocation of primary material and energy flows with corresponding emissions to relevant impact categories of each inventory item [165]. Inventory flows of the modelled systems were assessed through the LCA process using OpenLCA software with ReCiPe 2016 hierarchist method because this method translates the inventories on a global scale using 100-year time horizon for both mid and endpoint impact levels, while presenting the assessment results in numeric impact scores with equivalency unit for various impact categories [166-168]. 12 out of 18 midpoint and 14 out of 22 endpoint categories were selected. The excluded categories were water consumption, land use and resource depletion because these categories might not be applicable in other production systems with different geographical areas, while some categories were offset by environmental credits applied to avoided processes (e.g., avoided fertiliser production). Endpoint categories were also selected corresponding to the considered 12 midpoint categories.

2.4. Interpretation

As a final stage of the LCA study, interpretation process facilitates drawing conclusions by evaluating significance of objectives of the study and answering research questions based on results of the LCA. Comparison and aggregation of the assessment results can be performed during this stage, usually with endpoint results. Midpoint impact assessment focuses on the environmental burdens of a product system which are quantitatively analysed according to various midpoint impact categories, while endpoint impact assessment aggregates 18 categories into three endpoint impact categories to estimate final damages to human health, ecosystems and resource scarcity using common equivalency unit [56,167]. Any weighting or grouping of the selected 12 midpoint impact categories was not performed due to different units used in each category, while impact aggregation was performed for endpoint impacts according to impacts on human health and ecosystems with unit of disability adjusted life years (DALY), and species loss over time (species.yr), respectively [56].

3. Results

3.1. Midpoint impact assessment of renewable methane production

In this study, 11 system models developed for the seven production technologies were analysed according to 12 midpoint impact categories. As presented in Table 3, sewage sludge system (SEW-cen.ref) exhibited the highest global warming impact of 1.97 kg CO_2 eq/kg CH_4, mainly due to digestate dewatering in centrifuge (62.5% contribution to the impact), and more specifically, electricity consumption during centrifugation and methane emission from digestion in AED
with contribution of 49.4% and 24% to the impact, respectively. This was in agreement with another LCA study by Opatokun et al. [151] who reported over 65% contribution of centrifuge to the total impacts. Higher liquid content in the sewage sludge led to higher electricity consumption for dewatering by extending the process time.

The reference pig and dairy manure systems (P-lagoon.ref and D-lagoon.ref in Table 3) presented negative global warming impacts, which were greatly compensated by credits assigned to avoided emissions from the manure treatment in a conventional pond (around 61% of the total credit), and the avoided NPK fertilisers which also led to the lowest eutrophication impact of P-lagoon.ref. Although TS in dairy manure was slightly higher than that of pig manure, lower NPK contents in the dairy manure led to lower environmental credits than pig manure (-0.36 kg CO_2 eq/kg CH_4 and -0.4 kg CO_2 eq/kg CH_4 for dairy and pig manure, respectively). Since both reference systems (P-lagoon.ref and D-lagoon.ref) were modelled with lagoon and Geotube®, none of which required energy during operation, thereby the pre-processes for manure supply to the digester via pipeline and pre-treatment of manure (mixing with water) were found to be the major contributors to global warming impact with 48.3% and 48.5% contribution for pig and dairy manure, respectively. P-lagoon.ref system had the lowest impacts in 11 out of 12 categories except for terrestrial acidification because of emission of ammonia and nitrogen during field application of the NPK fertiliser. By contrast, the global warming impact of pig and dairy manure with AED (P-aed.ref and D-aed.ref in Table 3) ranked as the second and the fourth highest, mainly due to electricity demand from centrifuge operation. Unlike other systems, modelling of the landfill gas system (LF-mem.ref) began with landfill gas collection and transport without provision or pre-treatment of feedstock, which could be equivalent to biogas collection and transport to upgrading facility in other systems. This simpler production processes enabled less resource consumption with low emissions and leachates, which appeared to result in the lowest terrestrial acidification of the LF-mem.ref. Despite the reference wind system (W-pem.ref) had the fifth lowest global warming impacts, it appeared to be less competitive than the other systems because of the highest impacts in 4 out of 12 categories. Emissions of metals, metalloids and toxic chemicals during the construction of wind power station induced the highest human carcinogenic and non-carcinogenic toxicity impacts. For a similar reason, solar system (S-pem.ref) also had the highest impact in 4 out of 12 categories, with all four wind and solar systems (W-pem.ref, W-ae.ref, S-pem.ref, S-ae.ref) presenting considerably higher human carcinogenic and non-carcinogenic toxicity, and terrestrial ecotoxicity impacts than the other systems. The highest fine particulate matter formation of the solar system (S-ae.ref) was associated with the use of metals and minerals for manufacture of solar panels and AK electrolyser, as well as emissions of particulates and sulphur dioxide from energy consumption during the construction of the solar power station. The emission of sulphur dioxide also led to high terrestrial acidification of the solar system. Environmental credits earned from the CO2 by DAC, and use of waste heat from the methanation as a process heat for the DAC unit offset most of the global warming impacts of both wind and solar systems, with resulting impacts ranging relatively low among the 11 systems. Overall, wind and solar systems with AE had slightly lower impacts than PEM, due to lower energy consumption of AE (Table 2), but differences in the impacts were marginal (~ 1.4% lower impact than PEM on average). A study by Brandão et al. [170] also confirmed the minor difference in the impacts of PEM and AE, especially when renewable energy was used for the production.

Table 3 12 midpoint impacts of renewable methane production (per kg of renewable methane)

Midpoint impact category	Unit	Wind		Solar		Sewage sludge
		W-pem.ref ^{a}	W-ae.ref	S-pem.ref	S-ae.ref	SEW-cen.ref
Global warming	kg CO_2 eq	0.32	0.29	0.35	0.30	1.97^b
Fine particulate matter formation	kg $PM_{2.5}$ eq	9.4E-03	1.0E-02	1.0E-02	1.0E-02	1.8E-03
Human carcinogenic toxicity	kg 1,4-DCB	0.89	0.88	0.85	0.84	0.12
Human non-carcinogenic toxicity	kg 1,4-DCB	19.23	19.01	19.17	18.98	2.05
Ozone formation, Human health	kg NOx eq	1.4E-02	1.4E-02	1.5E-02	1.5E-02	3.1E-03
Freshwater ecotoxicity	kg 1,4-DCB	1.0E+00	1.0E+00	9.4E-01	9.3E-01	6.8E-02
Marine ecotoxicity	kg 1,4-DCB	1.310	1.308	1.25	1.23	0.09
Terrestrial ecotoxicity	kg 1,4-DCB	73.19	72.38	99.38	97.91	2.36
Freshwater eutrophication	kg P eq	7.2E-03	7.2E-03	7.4E-03	6.3E-03	1.5E-03
Marine eutrophication	kg N eq	4.3E-04	4.3E-04	4.9E-04	4.9E-04	2.3E-04
Terrestrial acidification	kg SO_2 eq	2.8E-02	2.8E-02	2.9E-02	2.9E-02	7.2E-03
Ozone formation, Terrestrial ecosystems	kg NOx eq	1.4E-02	1.4E-02	1.5E-02	1.5E-02	3.2E-03

Midpoint impact category	Pig manure		Dairy manure		Landfill gas	Food waste
	P-lagoon.ref	P-aed.ref	D-lagoon.ref	D-aed.ref	LF-mem.ref	FW-cen.ref
Global warming	-0.42	0.90	-0.33	1.16	0.34	0.94
Fine particulate matter formation	4.6E-04	1.8E-03	6.0E-04	2.2E-03	8.7E-04	9.2E-04
Human carcinogenic toxicity	0.03	0.10	0.03	0.12	0.24	8.3E-02
Human non-carcinogenic toxicity	0.26	1.68	0.39	2.08	1.79	1.13
Ozone formation, Human health	6.3E-04	2.8E-03	8.2E-04	3.3E-03	1.3E-03	1.5E-03
Freshwater ecotoxicity	1.8E-02	6.1E-02	2.4E-02	7.9E-02	1.1E-01	4.3E-02
Marine ecotoxicity	0.02	0.08	0.03	0.10	0.15	5.8E-02
Terrestrial ecotoxicity	0.63	2.50	1.22	4.25	9.38	1.71
Freshwater eutrophication	1.8E-04	1.2E-03	2.1E-04	1.3E-03	3.3E-04	8.0E-04
Marine eutrophication	-2.1E-05	1.7E-04	-1.5E-05	2.0E-04	5.3E-05	9.8E-05
Terrestrial acidification	2.0E-03	6.0E-03	2.4E-03	7.1E-03	1.8E-03	3.1E-03
Ozone formation, Terrestrial ecosystems	6.7E-04	2.9E-03	8.7E-04	3.4E-03	1.4E-03	1.5E-03

a. For description of system code (W-pem.ref, etc.) of each reference system, see Table S2 in Supplementary material.

b. Values in red and blue indicate the highest and the lowest impact among the 11 modelled systems.

3.2. End-point impact assessment of renewable methane production

The 14 analysed endpoint impact categories (five human impact and nine ecosystems impact categories) were aggregated according to endpoint damages on human health and ecosystem losses for the 11 production systems (Fig. 3). The wind and solar with PEM (W-pem.ref and S-pem.ref) ranked the highest in human health and ecosystems impacts, respectively, with all four wind and solar systems exhibiting significantly higher overall endpoint impacts than other systems. 51.6% of human health impacts of the W-pem.ref system were attributed to the highest human toxicity impacts, whereas the highest ecosystems impacts of the S-pem.ref system were mainly driven by freshwater eutrophication and ozone formation, terrestrial ecosystems impact with 30.4% and 12.4% of contribution, respectively (Table 4). Total human health impacts of wind and solar systems were in the order of W-pem.ref > S-pem.ref > S-ae.ref > W-ae.ref, and the ecosystems impacts were S-pem.ref > S-ae.ref> W-pem.ref > W-ae.ref (from high to low impact), which could be understood that systems with PEM had higher human health impacts than the systems with AE, while impacts on ecosystems were higher in solar systems than wind, regardless of the type of electrolyser.

Sewage sludge system (SEW-cen.ref) also had the highest impacts in 3 out of 14 categories with the highest global warming, human health impact. However, due to much lower impacts of the rest of the endpoint categories than other systems, the overall endpoint impacts of the sewage system were the fifth highest for both human health and ecosystems. Although both sewage sludge and food waste systems were modelled with the same processes (AED and centrifuge), much higher TS content in the food waste than the sewage sludge (27% vs. 6%) enabled the food waste to have higher yield, which resulted in 48.6% lower human health impacts and 52.6% lower ecosystems impacts than the sewage sludge system. Pig manure system (P-lagoon.ref) exhibited the lowest impacts in 13 out of 14 categories with the two lagoon systems (P-lagoon.ref and D-lagoon.ref) ranking the first and the second lowest in endpoint impacts, respectively, whereas the two AED systems (P-aed.ref and D-aed.ref) positioned the fifth and the sixth lowest for both impacts. Negative impact scores of global warming, human health impacts of the lagoon systems greatly reduced the total human health impacts, with about 87% reduction in P-lagoon.ref system and 52.4% reduction in D-lagoon.ref system. Despite the same feedstock of the systems, differences in the endpoint impacts could be explained by higher energy demand throughout the production process and different NPK contents affected by TS in the feedstock.

3.3. Scenario analysis based on key parameters

Scenarios were developed to confirm significance of the key parameters defined for each production system, which could be understood by changes in the global warming impacts caused by the parameter changes. The selected parameters for all 14 scenarios and the scenario analysis results are summarised in Table 5. Higher efficiency of wind and solar electricity presented greater reduction in global warming impact than increasing efficiency of electrolyser, with the impact reduction of 210% (scenario W1 in Table 5) and 94% (S1) when the wind and solar efficiency increased from 30% to 40% and from 16.5% to 19.5%, respectively. When the reference wind system (W-pem.ref) was modelled with 2020 Australian grid electricity (W1), the impact increased by almost 65 times (from 0.32 kg CO_2 eq/kg CH_4 to 20.8 kg CO_2 eq/kg CH_4), which confirmed importance of electricity source for impact assessment, as discussed in Reiter and Lindorfer [22]. Dewatering by Geotube® (SS1) had 78% lower impact than by centrifuge (SEW-cen.ref), which was also observed in the midpoint impact assessment of manure systems. When TS content in the sewage sludge decreased from 6% to 4% (from SEW-cen.ref to SS2), the impacts increased by 11.7%, possibly due to less nutrient recovery and more electricity consumption for digestate dewatering in SS2. Although TS content in the feedstock determines the biogas yield, digestible solid fraction in the TS, applied digestion technology and NPK contents in the TS also affect the yield [169], thus this scenario may only be applicable to systems with the same feedstock and digestion processes.

For the lagoon systems, concrete-constructed lagoon (P1) had 96% higher impact than HDPE-lined lagoon (P-lagoon.ref) although the impacts of both systems were still much lower than AED systems. Among all the selected parameters, increase in the distance of landfill gas collection and transport was found to have the greatest influence on the impact, with the impact increasing from 0.34 kg CO_2 eq/kg CH_4 to 2.87 kg CO_2 eq/kg CH_4 when the distance increased from 1 km to 8 km (from LF-mem.ref to LF4), and to 7.22 kg CO_2 eq/kg CH_4 with distances from 1 km to 20 km (LF5), which was also identified as the key factor for achieving a better environmental performance in the study by Tian et al. [47]. Biogas leakage by fugitive emissions and leakage of CH_4 during AD, gas transport (e.g., valves) and upgrading processes has been reported between 1-2% of the produced biogas [38,47,125]. With the 1% leakage applied for all reference 11 systems, 2% leakage was modelled as a scenario (LF3) which resulted in the impacts twice as high as the 1% leakage case (LF-mem.ref).

Despite the lower energy consumption (0.16 kWh/kg CH_4, in Table 1) of biogas upgrading by amine scrubbing (LF2) than the reference case by membrane separation (0.27 kWh/kg CH_4, LF-mem.ref), due to the chemical use (e.g., monoethanolamine) in the amine upgrading, the global warming impact was slightly higher than membrane upgrading (0.3% increase), whereas the upgrading by water scrubbing (LF1) performed the best among the three types of upgrading systems with 7% reduction in the impacts. Low TS content in the food waste (12%) with corresponding biogas yield of 0.03 m^3 per kg of food waste were modelled in scenario FW1 which was compared with the reference case with TS 27% and the yield of 0.07 m^3 per kg of food waste (FW-cen.ref). The low TS scenario demonstrated about 2.7 times higher impacts than the reference case because much higher liquid content in the food waste required more energy with longer operation time for dewatering, which resulted in 64.6% of contribution to the impacts.

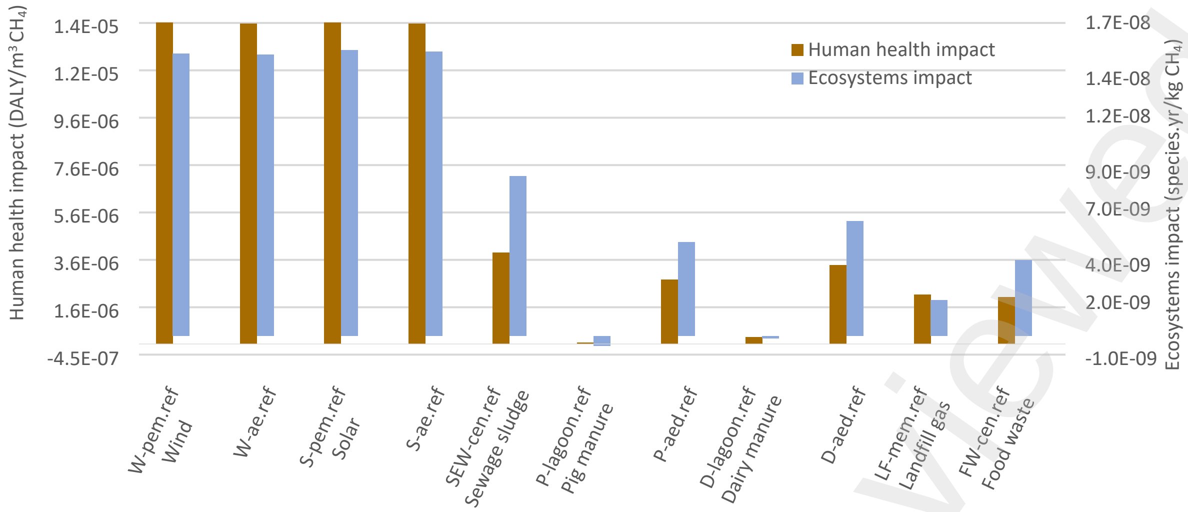

Fig. 3. Endpoint human health and ecosystems impact of renewable methane production

Table 4 Endpoint impacts of renewable methane production (per kg of renewable methane)

Endpoint impact category	Unit	Wind		Solar		Sewage sludge
		W-pem.ref [a]	W-ae.ref	S-pem.ref	S-ae.ref	SEW-cen.ref
Global warming, Human health	DALY	2.98E-07	2.52E-07	3.29E-07	2.88E-07	1.83E-06[b]
Human carcinogenic toxicity	DALY	2.93E-06	2.90E-06	2.78E-06	2.77E-06	4.00E-07
Fine particulate matter formation	DALY	5.90E-06	5.95E-06	6.38E-06	6.42E-06	1.15E-06
Human non-carcinogenic toxicity	DALY	4.11E-06	4.08E-06	4.06E-06	4.00E-06	4.67E-07
Ozone formation, Human health	DALY	1.27E-08	1.26E-08	1.35E-08	1.35E-08	2.86E-09
Terrestrial ecotoxicity	species.yr	8.40E-10	8.27E-10	1.13E-09	1.12E-09	2.70E-11
Freshwater ecotoxicity	species.yr	7.02E-10	6.97E-10	6.73E-10	6.72E-10	4.70E-11
Global warming, Terrestrial ecosystems	species.yr	9.01E-10	7.61E-10	9.95E-10	8.73E-10	5.52E-09
Freshwater eutrophication	species.yr	4.60E-09	4.59E-09	4.79E-09	4.23E-09	1.02E-09
Marine ecotoxicity	species.yr	1.40E-10	1.39E-10	1.32E-10	1.29E-10	9.57E-12
Global warming, Freshwater ecosystems	species.yr	2.46E-14	2.07E-14	2.71E-14	2.32E-14	1.51E-13
Ozone formation, Terrestrial ecosystems	species.yr	1.83E-09	1.81E-09	1.94E-09	1.93E-09	4.16E-10
Terrestrial acidification	species.yr	6.01E-09	6.01E-09	6.07E-09	6.09E-09	1.52E-09
Marine eutrophication	species.yr	7.34E-13	7.29E-13	8.32E-13	8.34E-13	3.99E-13

Endpoint impact category	Pig manure		Dairy manure		Landfill gas	Food waste
	P-lagoon.ref	P-aed.ref	D-lagoon.ref	D-aed.ref	LF-mem.ref	FW-cen.ref
Global warming, Human health	-3.89E-07	8.32E-07	-3.04E-07	1.07E-06	3.14E-07	8.69E-07
Human carcinogenic toxicity	9.58E-08	3.44E-07	1.15E-07	3.95E-07	8.05E-07	2.76E-07
Fine particulate matter formation	2.93E-07	1.15E-06	3.76E-07	1.37E-06	5.45E-07	5.75E-07
Human non-carcinogenic toxicity	5.87E-08	3.83E-07	8.84E-08	4.74E-07	4.09E-07	2.57E-07
Ozone formation, Human health	5.71E-10	2.57E-09	7.42E-10	3.02E-09	1.23E-09	1.32E-09
Terrestrial ecotoxicity	7.20E-12	2.85E-11	1.39E-11	4.85E-11	1.07E-10	1.96E-11
Freshwater ecotoxicity	1.22E-11	4.20E-11	1.65E-11	5.45E-11	7.80E-11	3.01E-11
Global warming, Terrestrial ecosystems	-1.17E-09	2.51E-09	-9.14E-10	3.24E-09	9.49E-10	2.62E-09
Freshwater eutrophication	1.20E-10	7.75E-10	1.40E-10	8.40E-10	2.19E-10	5.36E-10
Marine ecotoxicity	2.37E-12	8.49E-12	3.24E-12	1.10E-11	1.54E-11	6.05E-12
Global warming, Freshwater ecosystems	-3.20E-14	6.86E-14	-2.50E-14	8.84E-14	2.59E-14	7.16E-14
Ozone formation, Terrestrial ecosystems	8.68E-11	3.77E-10	1.12E-10	4.42E-10	1.80E-10	1.92E-10
Terrestrial acidification	4.24E-10	1.28E-09	5.07E-10	1.51E-09	3.76E-10	6.55E-10
Marine eutrophication	-3.53E-14	2.93E-13	-2.60E-14	3.37E-13	9.12E-14	1.65E-13

a. For description of system code (W-pem.ref, etc.) for each reference system, see Table S2 in Supplementary material.

b. Values in red and blue indicate the highest and the lowest impact among the 11 modelled systems.

Table 5 Scenario analysis based on 14 parameters

	Scenario	Parameter 1	Parameter 2	Parameter 3	Impact (kg CO_2 eq/ kg CH_4)	Changes in impact (kg CO_2 eq/ kg CH_4)	Changes in impact (%)
		Conversion efficiency	Electrolyser efficiency	CO_2 capture			
Wind	W-pem.ref [a]	Wind (30%)	PEM (61.8%)	Direct air capture (DAC)	0.32		
	W1	Wind (high, 40%)[b]			-0.35[c]	-0.68	-210
	W2	2020 Australian grid			20.8	20.48	6380
	W3	Wind (30%)	PEM (high, 70%)		0.029	-0.29	-91
	W4		PEM (low, 54%		0.681	0.36	113
Solar	S-pem.ref	Solar 16.5%	PEM (61.8%)	Direct air capture (DAC)	0.35		
	S1	Solar (high, 19.5%)			0.02	-0.33	-94
		Total solid (TS)	**Digester type**	**Dewatering**			
Sewage sludge	SEW-cen.ref	6%	AED	Centrifuge	1.97		
	SS1			Geotube®	0.43	-1.55	-78
	SS2	4%	AD	Centrifuge	2.20	0.23	12
		Digester type	**Lagoon material**	**Dewatering**			
Pig manure	P-lagoon.ref	CAL	HDPE	Geotube®	-0.42		
	P1		Concrete	Geotube®	-0.02	0.40	96
		LFG collection distance	**Biogas leakage**	**Biogas upgrading**			
Landfill gas (LFG)	LF-mem.ref	1 km	1%	Membrane	0.34		
	LF1			Water scrubbing	0.31	-0.02	-7
	LF2			Amine scrubbing	0.34	0.001	0.3
	LF3	1 km	2%	Membrane	0.68	0.34	100
	LF4	8 km	1%	Membrane	2.87	2.54	744
	LF5	20 km			7.22	6.88	2023
		Total solid (TS)	**Digester type**	**Dewatering**			
Food waste	FW-cen.ref	27%	AD	Centrifuge	0.94		
	FW1	12%			2.52	1.58	168

a. For description of reference systems' system code (W-pem.ref a, etc.), see Table S2 in Supplementary material.
b. Parameters that are used in the scenario analysis are expressed as underlined.
c. Numbers in red and blue indicate higher (increasing) and lower (decreasing) impact values than reference system, respectively, while impact values of reference systems are presented in black colour.

4. Discussion

4.1. Comparison of impact assessment results with conventional natural gas production

Midpoint impacts of the 11 modelled systems were compared with natural gas production to understand strength and weakness in various environmental concerns. The natural gas system included gas extraction and processing (e.g., sulphur and moisture removal) where the extracted gas was purified up to the pipeline standard. As presented in Table 6, the natural gas system ranked eighth in global warming impacts among the 12 production systems (11 renewable methane and one natural gas production systems) with 0.79 kg CO_2 eq/ kg CH_4, mainly due to methane leakage (0.5% of the produced natural gas during gas extraction and processing) and partial flaring of the gas. However, the natural gas system outperformed in five out of the 11 lowest impacts from the reference pig manure system (P-lagoon.ref), which resulted in the pig system having the six lowest and the natural gas system having the five lowest midpoint impacts (Table 7). The lowest terrestrial ecotoxicity impact of the natural gas system was attributed to metal emissions from electricity consumption throughout the gas processing processes (26.7% of contribution rate) and construction of gas processing plant (20% of contribution rate). Despite the impact compensation by avoided fertiliser production for the reference pig manure case, much greater emissions of copper from biogas upgrading facility led to the second lowest impact of the pig manure case. Contribution of the electricity consumption was greater in both freshwater and marine ecotoxicity impacts of the natural gas system with around 52% contribution rate for both impacts, but the impacts were still the lowest among the modelled systems due to low water emissions. Overall, the natural gas system had the lowest impacts in human toxicity, ecotoxicity and freshwater eutrophication categories, with other impact categories including fine particulate matter formation, marine eutrophication, terrestrial acidification, and ozone formation, terrestrial ecosystems impacts were the third lowest.

Table 6 Rank of global warming impact of renewable methane and natural gas production systems

Source of methane (gas) production	System code[a]	Global warming impact (kg CO$_2$ eq/ kg CH$_4$)[b]	Rank[c]
Pig	P-lagoon.ref	-0.42	1
Dairy	D-lagoon.ref	-0.33	2
Wind	W-ae.ref	0.29	3
Solar	S-ae.ref	0.3	4
Wind	W-pem.ref	0.32	5
LFG	LF-mem.ref	0.34	6
Solar	S-pem.ref	0.35	7
Natural gas		0.79	8
Pig	P-aed.ref	0.9	9
Food	FW-cen.ref	0.94	10
Dairy	D-aed.ref	1.16	11
Sewage	SEW-cen.ref	1.97	12

a. For system description of each reference system (e.g., W-pem.ref), see Table S2 in Supplementary material.

b. Impact is expressed as per kg of the produced renewable methane or natural gas. Both gases are assumed to have the same gas quality (e.g., methane content > 97%) which are suitable for grid injection.

c. Rank is in the order of the best environmental performance to the worst performance (e.g., rank 1 indicates the best performance).

Table 7 Midpoint impact comparison of renewable methane and natural gas production

Midpoint impact category	Natural gas[a]	Wind		Solar		Sewage
		W-pem.ref	W-ae.ref	S-pem.ref	S-ae.ref	SEW-cen.ref
Global warming (kg CO$_2$ eq)	0.79	0.32	0.29	0.35	0.30	1.97
Fine particulate matter formation (kg PM$_{2.5}$ eq)	7.3E-04	9.4E-03	1.0E-02	1.0E-02	1.0E-02	1.8E-03
Human carcinogenic toxicity (kg 1,4-DCB)	0.02	0.89	0.88	0.85	0.84	0.12
Human non-carcinogenic toxicity (kg 1,4-DCB)	0.26	19.23	19.01	19.17	18.98	2.05
Ozone formation, Human health (kg NOx eq)	7.5E-04	1.4E-02	1.4E-02	1.5E-02	1.5E-02	3.1E-03
Freshwater ecotoxicity (kg 1,4-DCB)	0.01	1.02	1.01	0.94	0.93	0.07
Marine ecotoxicity (kg 1,4-DCB)	0.01	1.31	1.31	1.25	1.23	0.09
Terrestrial ecotoxicity (kg 1,4-DCB)	0.16	73.19	72.38	99.38	97.91	2.36
Freshwater eutrophication (kg P eq)	9.0E-05	7.2E-03	7.2E-03	7.4E-03	6.3E-03	1.5E-03
Marine eutrophication (kg N eq)	9.5E-06	4.3E-04	4.3E-04	4.9E-04	4.9E-04	2.3E-04
Terrestrial acidification (kg SO$_2$ eq)	2.0E-03	2.8E-02	2.8E-02	2.9E-02	2.9E-02	7.2E-03
Ozone formation, Terrestrial ecosystems (kg NOx eq)	9.2E-04	1.4E-02	1.4E-02	1.5E-02	1.5E-02	3.2E-03

Midpoint impact category	Pig manure		Dairy manure		Landfill gas	Food waste
	P-lagoon.ref	P-aed.ref	D-lagoon.ref	D-aed.ref	LF-mem.ref	FW-cen.ref
Global warming (kg CO$_2$ eq)	-0.42	0.90	-0.33	1.16	0.34	0.94
Fine particulate matter formation (kg PM$_{2.5}$ eq)	4.6E-04	1.8E-03	6.0E-04	2.2E-03	8.7E-04	9.2E-04
Human carcinogenic toxicity (kg 1,4-DCB)	0.03	0.10	0.03	0.12	0.24	0.08
Human non-carcinogenic toxicity (kg 1,4-DCB)	0.26	1.68	0.39	2.08	1.79	1.13
Ozone formation, Human health (kg NOx eq)	6.3E-04	2.8E-03	8.2E-04	3.3E-03	1.3E-03	1.5E-03
Freshwater ecotoxicity (kg 1,4-DCB)	0.02	0.06	0.02	0.08	0.11	0.04
Marine ecotoxicity (kg 1,4-DCB)	0.02	0.08	0.03	0.10	0.15	0.06
Terrestrial ecotoxicity (kg 1,4-DCB)	0.63	2.50	1.22	4.25	9.38	1.71
Freshwater eutrophication (kg P eq)	1.8E-04	1.2E-03	2.1E-04	1.3E-03	3.3E-04	8.0E-04
Marine eutrophication (kg N eq)	-2.1E-05	1.7E-04	-1.5E-05	2.0E-04	5.3E-05	9.8E-05
Terrestrial acidification (kg SO$_2$ eq)	2.0E-03	6.0E-03	2.4E-03	7.1E-03	1.8E-03	3.1E-03
Ozone formation, Terrestrial ecosystems (kg NOx eq)	6.7E-04	2.9E-03	8.7E-04	3.4E-03	1.4E-03	1.5E-03

a. The natural gas production system was modelled based on onshore natural gas extraction and processing from Ecoinvent database v3.8 [57] with 0.5% of the produced gas being lost as a gas leakage and 5% of the gas being flared. Gas extraction was assumed to be performed by hydraulic fracturing (fracking).

b. Values in red and blue indicate the highest and the lowest impact among the 12 modelled systems.

4.2. Factors affecting yield and environmental impacts

Since environment impact assessment results are usually expressed in per unit of produced renewable methane or biogas, changes in the yield would greatly affect degree of the impacts. This section discusses various factors that affect the yield and the impacts. The methane yield from AD of waste material can be affected by its carbon-nitrogen ratio (C/N ratio) with the optimal ratio between 15-30:1 [55,133,140,171]. The ratio lower than the optimal can increase pH of the digestion process and lead to ammonia accumulation which can hinder activity of microorganisms and ultimately reduce the biogas yield [133,172]. C/N ratio differs according to feedstock, with animal manures having relatively low carbon content than agricultural residues [173], hence co-digestion, a simultaneous digestion of two or more feedstocks, consisting of low-carbon feedstock with carbon-rich feedstock, has been suggested in a number of studies for higher yield of biogas [55,133]. Oh et al. [174] studied the synergetic effect of co-digestion of food waste and wood chip, and found 6.4 times higher methane yield than mono-digestion of food waste, with food waste to wood chip ratio of 0.5 :1 presented as optimal. Optimal ratio of food waste and wheat straw of 9:1 was proposed in another study by Shi et al. [175], who confirmed increased biogas yield by 67% and 50% when compared with mono-digestion of food waste and wheat straw, respectively. It should be noted that in order to maximise the yield, considering a single factor (e.g., C/N ratio) may not be adequate to determine the feed mixture with optimal mixture rate for co-digestion [169], therefore, other factors, such as digester design, solid content of feedstock and climate factors should also be considered.

The biogas yield is determined by a function of various factors, such as volatile solid (VS), biodegradability of feedstock, loading rate and hydraulic retention time [133,161]. Decomposition rate of VS with respect to concentration of proteins, carbohydrates and lipid in the feedstock affect the biogas yield, with theoretical methane yield of 415 L CH_4/ kg VS and 1014 L CH_4/ kg VS from carbohydrate and lipid content, respectively [23]. As discussed in the scenario analysis, low TS content in the food waste resulted in low biogas yield and higher environmental impacts. However, given the source-separated food waste has TS about 25.7%, while feedstocks with up to 25% of TS can be treated by AD [125,161], there is a great potential for increasing the yield by supplying high TS food waste to digester without mixing with water.

Digestion temperature affects activity of anaerobic bacteria in the digester that decomposes organic matter to produce biogas with neutral pH and mesophilic conditions (temperatures around 35 °C) known to be optimal conditions for the digestion [133,140]. Majority of currently operating digesters are designed to operate within the temperature ranges between 20-50 °C, but CAL, the dominant type of digester in Australia, is installed without heat supply, and operates in conditions with ambient temperatures between 10-30 °C. However, temperate to subtropical climate (annual average temperature between 5-30 °C) in south-eastern coastal areas in Australia where most biogas facilities are located (Fig. 4) can be favourable to adopt waste-based renewable methane production systems [169,176].

Generally, AD systems installed with heat supply and agitator (e.g., continuous stirred tank reactors) are know to have higher biogas yields by shorter retention time in the digester with less interference of ambient air temperature compared to conventional lagoon and unheated digesters, with around 30% higher biogas yield in mesophilic than psychrophilic digestion conditions (10-30 °C) [23,32,169,177], although heated digesters exhibit operational complexity and higher capital and operating costs with more energy requirement which can ultimately lead to higher environmental impacts [169]. Influence of climatic factors was also discussed in the landfill gas study by Karanjekar et al. [145], who discovered higher production potential of renewable methane from landfill with high temperature and humidity (and thus higher moisture content in the waste) conditions.

As presented in the scenario analysis, efficiency of wind electricity appears to be an important factor for LCA studies. The amount of wind electricity can be affected by wind condition of the area where the wind power station is located, and the wind condition determines selection of the wind turbine [67]. The wind conditions in the Musselroe wind farm (TAS, Australia) are classified according to the International Electrotechnical Commission standard as high, medium and low wind classes with annual average wind speed of 8.5-10 m/s, 8.5-7.5 m/s and 7.5-6.5 m/s, respectively. According to the farm, 24.5% more electricity is generated in the high wind than the medium wind class [67]. Since influence of production capacity of renewable electricity on overall impacts is reported as insignificant [19,170], the impacts are more likely to be affected by the efficiency of wind electricity.

Environmental credits assigned to avoided emissions from substitution of NPK fertiliser and manure management in conventional effluent pond are also reported in other studies as a major factor affecting the overall impacts with contribution of 22% and 40% to global warming impact of pig manure system [131]. Traditionally, manure and digested sludge have often been treated in an open lagoon (for liquid slurry) or stockpiled (for solid manure) with the open lagoon being reported as a dominant emission source of biogas production system [32,55,169]. The methane conversion factor (Cf_{CH4}) of the manure systems in this study is for an open effluent pond with Cf_{CH4} of 0.75, but when stockpiling (Cf_{CH4} of 0.02) was assumed, the avoided emissions would decrease from -0.85 kg CH_4 to -0.02 kg CH_4 per kg of the produced renewable methane, which indicates less environmental credits.

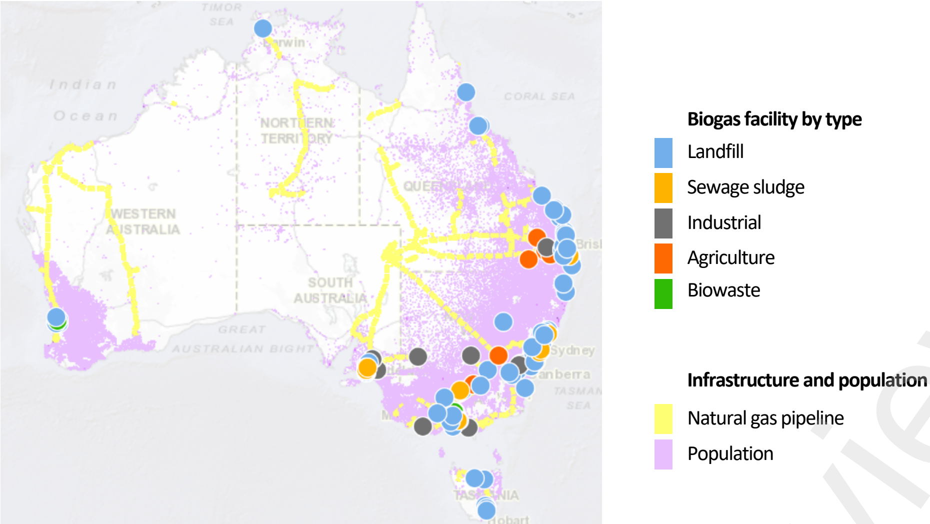

Fig. 4. Distribution of natural gas pipelines, population and biogas facilities
* Map is constructed using NationalMap tool [178].

4.3. Factors affecting market expansion

While Carbon Farming Initiative acted as a foundation for accelerated development of biogas sector in Australia during 2011-2014 [127], other numerous government supports exist for promoting renewable energy production, such as Emissions Reduction Fund (ERF), which provides financial support for renewable energy sold to the grid [64]. More recently, Biomethane Method Package Variations by the Australian Government came into effect in January 2022 to acknowledge the value of biogas and biomethane as a substitute for natural gas with the potential of achieving net zero emissions. This new framework accredits biomethane production from waste materials, including wastewater, landfill gas and animal manure [179]. Despite the existing support, only 12 manure-based biogas projects were registered in the ERF (as in Oct 2022), and only 0.09% and 0.42% of the total electricity in 2019-20 was produced by biogas from sewage sludge and landfill gas, respectively, with almost no increase in the production since 2018 [63,127,180]. Together with implementation of these carbon reduction initiatives, expected increase of natural gas price coincides with the increasing demand for gas. The risks in the gas supply due to higher seasonal fluctuation of the gas would emphasise replanning of the national energy mix, possibly by increasing renewable energy production [16,181,182]. Expected increase in production capacity of renewables with lower production cost and retirement of coal power stations may also encourage the production of renewable energy [16].

Due to commercial immaturity of renewable methane, specifications and standards for ensuring quality of the methane for grid injection are not available yet. Instead, relevant gas standards, such as Specifications for General Purpose of Natural Gas (AS 4564), Gas Installation (AS 5601) and Gas Distribution Networks (AS 4645) are used by the Australian industries [36,183,184]. Gas quality issues have been reported as some of the obstacles that hinder renewable market expansion [24]. This is important because the specifications determine the selection of biogas upgrading technology which has different levels of resource requirements, cost, operation pressure and impurity removal rates (see Table 1 for biogas upgrading). Removal of hydrogen sulphide and moisture content is important to prevent corrosion in the gas pipeline and reduction in energy value of the gas [23,24]. For pipeline transmission of the renewable methane, it should be compressed up to the pressure threshold of gas pipeline with the pressure greater than and lower than 10.5 bar for high- and low-pressure pipelines, respectively [184].

One of the greatest potentials of renewable methane is its transmission via natural gas pipelines. However, given there is no grid-connected renewable methane facility in Australia, network connection from the methane production facility to distribution mains with some modifications for existing infrastructures may be required. While for more efficient production, new biogas facilities may consider factors such as spatial distribution of demand and supply of the gas, possibility of integrating already existing infrastructures and pipelines, and geographical proximity of each component of the production system (Fig. 4) [158]. A feasibility study before the construction of the methane production facility could also be beneficial as exampled in Veiga et al. [146] who analysed feasibility of biomethane production based on the distance of gas pipelines in Brazil. The study concluded that the distance of pipeline for biomethane distribution up to 50 km could be practical considering the current network system in Brazil.

4.4. Challenges and opportunities based on case study

4.4.1. Renewable methane potential from a piggery farm

A case study was performed for Bears piggery farm (VIC, Australia) in order to understand challenges and opportunities for the methane production sector. The farm is capable of producing 3,350 m^3 of biogas per day in a CAL with 18 ML treatment volume[130]. Approximately 23,000 SPU (standard pig unit where one SPU is equivalent to 1 grower pig of 40 kg nominal liveweight) produces about 4,340 kg VS per day that is fed to a lagoon with loading rate of 0.24 kg VS/m^3 and hydraulic retention time (HRT) of 36 days [130]. The environmental credits were included the same as in the reference cases.

Based on the methane yield of 0.33 kg CH_4/kg VS of the farm, 518.7 t of renewable methane can be produced per year with emission savings of 286 t CO_2 eq, which is 23.6% higher emission savings per year than the reference case due to higher methane yield of this farm than the yield of 0.22 kg CH_4/kg VS for the reference case. The lower yield estimate may be more realistic because the lagoon system, especially still-operating floating cover lagoon, has lower yield than engineered digester due to biogas loss and limitations in collectable biogas (e.g., 63% of the total produced biogas can be collected from the floating cover lagoon) [24]. However, this could be improved for newly-constructed and completely-sealed lagoons which are equipped with sealed biogas collection pipes and vales, and biogas and sludge accumulation monitoring units [61].

There are currently 579 piggery farms in Australia with a total of 2,577,597 SPUs [185,186]. When 90% of total manure is assumed to be collectable [127,169], the Australian piggery sector could have a theoretical renewable methane yield of 68,900 t/a (estimated by VS production rate of 90 kg VS/SPU/year [62] with yield potential of 0.33 kg CH_4/kg VS (yield same as the Bears piggery farm), which is equivalent to 38 kt of emission savings per year, and energy equivalent to 3,445 TJ/a by applying heating value of 50 MJ/kg CH_4. Type and frequency of manure collection is important to avoid solid content loss because considerable amount of VS in the manure can be lost with unsuitable manure cleaning and frequency which greatly affect the biogas yield and recoverable amount of NPK content [169]. Risk management for operation of CAL is also important for early detection of gas leaks to prevent corrosions, and on-time removal of the accumulated sludge to avoid reductions in biogas yield and blockage in sludge pipes. 5-year time interval of sludge removal in this study is determined by sludge accumulation rate of 0.00137 m^3 sludge/kg VS which is in the low range of values reported by other studies [23,61,130,133]. When higher rate (0.003 – 0.0094 m^3/kg VS) is applied, more frequent sludge removal should be performed.

4.4.2. Opportunities in CO_2 capture

While this study only considered direct air capture (DAC) as a source of CO_2 for methanation, other types of CO_2 sources, such as CO_2 captured from cement plant or fossil power station have been reported in other studies [18,20]. However, the CO_2 of these sources is from fossil origin, and transport of the CO_2 may not be economically viable nor sustainable because construction of the transport pipelines requires considerable amount of resources, materials and associated costs [187]. Consumption of water, energy and sorbent from the DAC process has been a subject of discussion, which is also closely related to production cost. As modelled in this study, waste heat from methanation can be used as process heat, or the DAC unit can be integrated with renewable energy plant. Also, DAC is still in the early stage of commercialisation, and it is expected to become more cost-effective by increasing production scale with better efficiency [76,115]. According to Climeworks, 7.5 kg of sorbent is required per tonne of captured CO_2 for 4 kt capacity DAC plant, which can be reduced to 3 kg/t captured CO_2 for plant with 100 kt capacity [77].

Conclusions

This study evaluated the environmental impacts of seven different renewable methane production technologies based on 11 system models with 14 alternative scenarios, developed using various data sources from literature and commercial data, as well as data from manufactures, operators and case studies. Despite the integration of renewable electricity into the production system, the renewable methane production from wind- and solar-powered electrolysis with methanation had higher overall impacts among the 11 system models, with particularly high human toxicity and ecotoxicity impacts, which were mainly attributed to material and energy flows for construction of wind and solar power stations. However, environmental credits applied to air-captured carbon dioxide for methanation and recycling of waste heat from the methanation can offset most of the global warming impacts of the systems resulting in considerably lower impacts for the wind and solar systems. Global warming impact was the highest in the renewable methane production by anaerobic digestion of sewage sludge, followed by dairy manure, due to electricity consumption for digestate dewatering by centrifuge with contribution of 62.5% and 69.2% to the impact, respectively. The centrifuge process was one of the major contributors to global warming impact with contribution rate between 62.5-70%, which was exacerbated by low total solid (TS) content in the sewage sludge systems, with 11.7% higher impact in the low TS system (4%) than higher TS (6%) system due to higher energy consumption during centrifugation. Negative global warming impacts, -0.42 kg CO_2 eq/kg CH_4 and -0.33 kg CO_2 eq/kg CH_4, were found in the pig and dairy manure in lagoon and Geotube® systems, respectively, which were achieved by environmental credits gained by avoided emissions from conventional manure management and substitution of fertiliser, as well as no electricity consumption during operation of the lagoon and Geotube®. The highest endpoint human health impacts of the reference wind system (W-pem.ref) was mainly associated with human toxicity impacts with 51.6% contribution, while freshwater eutrophication, and ozone formation, terrestrial ecosystems were found to be the most impactful categories leading to the highest ecosystems impact of solar system (S-pem.ref) with 30.4% and 12.4% of contribution, respectively.

Scenario analysis found that increasing distance from 1 km to 8 km for transport of the collected landfill gas to biogas upgrading facility had the greatest influence on the assessment result with 8.4 times higher global warming impacts. Changes in efficiency of wind and solar electricity generation also appeared to be one of the key parameters, with 210% and 94% decrease in the impacts when the wind and solar efficiency increased from 30% to 40% and from 16.5% to 19.5%, respectively. Among three types of biogas upgrading technologies, water scrubbing performed the best with 7% impact reduction than membrane upgrading, whereas difference in the impact of membrane and amine scrub was insignificant. When midpoint impacts of the 11 modelled renewable methane systems were compared with conventional natural gas production, the natural gas system ranked eighth in global warming impacts with the lowest impacts in five out of the 12 midpoint impact categories.

Author Contributions

Hannah Hyunah Cho: Conceptualization, writing – original draft, data curation, formal analysis. **Vladimir Strezov:** Conceptualization, methodology, writing – review & editing, supervision, validation. **Tim J. Evans:** supervision, validation, writing – review & editing.

Conflicts of interest

There are no conflicts to declare.

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