Australia's National
Science Agency

Methane contributions from holding ponds – a desktop study to identify emissions potential and controls in CSG holding ponds and other aquatic systems in Queensland

Se Gong, Richard Schinteie, Emma Crooke, Linda Stalker and David J. Midgley

Report number: EP2023-2256
09/06/2023

Citation

Gong, S., Schinteie, R., Crooke, E., Midgley, D.J. (2023) Methane contributions from holding ponds – a desktop study to identify emissions potential and controls in CSG holding ponds and aquatic systems in Queensland. CSIRO, Australia. Report number: EP2023-2256

Contents

Figures

Tables

Acknowledgments

This research has been funded through CSIRO's Gas Industry Social and Environmental Research Alliance (GISERA) with contributions from the Australian Government's Department of Industry, Science and Resources. GISERA is a collaboration between CSIRO, Commonwealth, state and territory governments and industry established to undertake research on the impacts of onshore gas exploration and development on the environment, and the socio-economic costs and benefits. For information about GISERA's governance structure, projects and research findings visit https://gisera.csiro.au

We thank Faiz Mohinudeen and Charles Heath for CSIRO internal report review.

We acknowledge the Traditional Custodians whose ancestral lands we live and work upon and we pay our respects to their Elders past and present. We acknowledge and respect the deep spiritual connection and the relationship that Aboriginal and Torres Strait Islander people have to Country.

Executive summary

Methane is the second most important greenhouse gas in terms of its contribution to increasing global temperatures. Indeed, quantification and reporting of emissions has become the focus area of many governments, including Australia, in an effort to reach a net of zero emissions across society. Australia specifically aims to achieve net zero emissions by 2050 through a mix of both regulation and action by the public and private industry. The Glasgow Climate Pact, of which Australia is a signatory, has committed to reduce methane emissions by 30% by 2030, and this project is a first step in identifying potential sources and mitigation strategies to reach that goal as part of the Global Methane Pledge.

It is noteworthy that some emission processes and compositions are highly variable and their contribution to some emissions are poorly understood making quantifiable estimates challenging and contentious. Furthermore, compared with point sources, diffuse sources of emissions tend to be somewhat harder to measure, track and verify.

Coal seam gas (CSG) holding ponds in Queensland have been identified as a significant source of this uncertainty, as little is known about their emission potential (particularly as diffuse sources). Under current rules, diffuse emissions from CSG holding ponds, where the methane is generated in the pond due to biological activities, are not regarded as 'fugitive' emissions. By definition, fugitive emissions come from operational infrastructure such as vents, flue gases, pipelines etc. Emissions from water holding ponds, while being part of the infrastructure for CSG, are not regarded as fugitive. This study attempts to use literature from the CSG industry and natural analogues to better estimate if such emissions are significant and how they should be included in greenhouse gas estimations. The data presented in this report is open-file and derived from readily accessible information.

This study evaluates whether CSG water holding ponds could be a significant source of methane emissions. As there are limited data on emissions from CSG water holding ponds, the project seeks to leverage existing information on the relative contributions and key controls on the rates of emissions from aquatic systems such as lakes, ponds and reservoirs.

Review of natural and anthropogenically constructed waterbodies

This study reviewed methane emissions and water chemistry data from natural and anthropogenically constructed waterbodies (e.g., ponds or dams) as well as methods for measuring methane emissions published in peer-reviewed journals.

It is important to note that the literature is somewhat challenging to synthesise, for example, not all studies on emissions necessarily include water chemistry data, and vice versa. Furthermore, the types of emissions reported often vary, with studies variously presenting ebullitive versus diffusive emissions. Despite this, it was possible to group some key concepts from this literature review, these are summarised below:

- There is considerably more emissions data from waterbodies from temperate climates compared to subtropical and tropical climates in the literature.

- Ebullition (bubbling) and diffusion (dissolved methane in water) are the dominant pathways of methane emissions, but the relative contributions of these different pathways vary widely (by up to two orders of magnitude) and is influenced by a range of location-specific, biotic and abiotic factors.
- Methane emissions, especially ebullitive, from natural and anthropogenically constructed waterbodies are positively correlated with increasing climatic temperature, for example, methane emissions from waterbodies in subtropical and tropical regions are higher than those from temperate or arctic regions.
- Small and shallow waterbodies tend to emit more methane, especially via the ebullitive pathway. This is related to residency time that individual bubbles have in travelling to the ponds surface.
- Nutrients, such as total phosphorus and dissolved organic carbon, are positively correlated with methane emissions.
- Depth of waterbodies is negatively correlated with ebullitive methane emission but have no obvious relationship with diffusive methane emissions.

The review of sampling and analysis methods for methane emission measurements from aquatic systems highlighted various limitations and advantages of different approaches.

- Funnel traps and/or floating chambers, in conjunction with headspace gas analysis of dissolved methane in surface water environments are commonly used to measure emissions. These methods may overestimate or underestimate the emissions due to localised ebullition in the waterbody.
- Eddy covariance measurement can provide estimates of flux over a wide area, and capture variation over long timescales, however, the required equipment and data-processing are somewhat intensive, especially for long time periods. This may not be feasible for multiple ponds, particularly if multiple sensors are used in order to adequately sample the environment.
- Mobile instrumentation, such as cavity ring down spectroscopy (CRDS) or off-axis integrated cavity output spectroscopy (OA-ICOS) based sensors, may be a portable solution for making measurements in the field over shorter time scales, but unless they are integrated with eddy covariance methods, the usefulness of the results may be limited by shorter periods on site and limited coverage of the surface of the water body.
- Autonomous surface vehicles (ASVs) and unmanned aerial vehicles (UAVs), in conjunction with newer, lightweight instrumentation may provide an opportunity to identify methane hotspots in smaller ponds, however, this is highly specialised equipment and may not be suitable for larger ponds over long-time scales.
- High resolution satellite sensing allows investigation of methane emissions from small areas. Quantifying methane emissions, however, is still limited by the detection limit of the satellite sensors which make the detection of relatively low diffusive source more challenging.

Existing knowledge on CSG holding ponds

The review of emissions, water chemistry and microbiological data from CSG water holding ponds, mainly including produced water ponds and brine ponds revealed there is a dearth of information on methane emissions from small ponds associated with CSG activities:

- There are a few measurements on methane emissions from CSG holding ponds in Queensland.
- There are only a small number (<10) isolated microbes grown in the laboratory on culture media which do not assist in understanding the diversity of microbial communities in CSG holding ponds or their function as a community.
- Water chemistry of produced water and brine is different between the Surat and Bowen basins. Water chemistry between produced water and brine in the same basin, however, are similar.
- There is no information on organic carbon in the sediments in the CSG holding ponds. This could be a significant pool of carbon for microbes.

Emission estimation from natural analogues

At the core of this project, natural and anthropogenically constructed waterbodies were used as analogues for the estimation of methane emissions from CSG holding ponds in Queensland. Methane emissions from these small to medium sized waterbodies (< 100 ha) together with their climate zones were mainly considered in the below scenarios.

Scenario 1 assumed that the CSG holding ponds in Queensland are analogues of subtropical waterbodies. Estimated methane emissions from CSG holding ponds is around **30 mg/m^2/d**.

Scenario 2 assumed that the CSG holding ponds in Queensland are analogues of temperate waterbodies. Estimated methane emissions from CSG holding ponds is around **15 mg/m^2/d**.

Scenario 3 assumed a mixture, in which 50% of the CSG holding ponds are in temperate regions, while the other 50% are in subtropical regions of Queensland. Estimated methane emissions from CSG holding ponds is around **22 mg/m^2/d**.

One of the key assumptions for the above scenarios is that waterbodies, like ponds, lakes or dams are sufficient analogues to CSG holding ponds in terms of emissions, water chemistry and microbiology, but as discussed below, there are reasons to believe these emissions estimates may significantly underestimate the true emissions generated from CSG holding ponds.

Firstly, it is evident that total phosphorous and dissolved organic carbon contents are up to an order of magnitude higher in the CSG produced water ponds and two orders of magnitude higher in brine ponds compared to natural and anthropogenically constructed waterbodies. These nutrients, at least, would likely further enhance rates of methanogenesis (methane-generating), but may also enhance rates of methantrophy (methane-eating).

Secondly, different to natural and anthropogenically constructed waterbodies, an extremely high content of inorganic carbon in CSG holding ponds could be another large pool of carbon as a source for methane generation.

Thirdly, twenty-two CSG holding ponds could not be included in these estimates because the area information of these ponds is not publicly available. Further, the CSG ponds collated in this report

may not be all of the CSG holding ponds in Queensland which is limited by publicly available resources.

Fourthly, the definition of "temperate" and "subtropical" as the literature describe it vary. For example, most climatic maps of Australia describe Sydney as being in a "temperate zone", however, many cities in the central USA described as having "temperate climates" have notably colder temperature.

Hence, it is obvious that the estimates in the scenarios using natural and anthropogenically constructed waterbodies as analogues are conservative. This is further reflected from two examples related to CSG holding ponds in Figure 1 with other contexts for comparison.

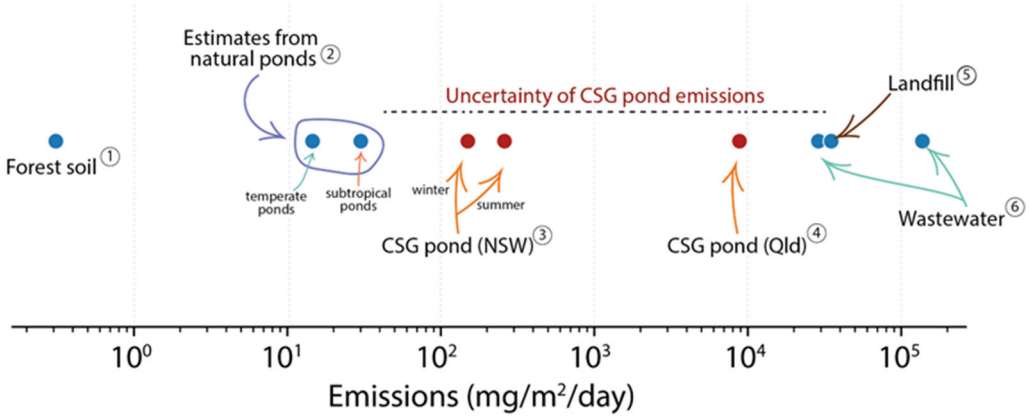


Figure 1 A diagram showing methane emissions from the estimates of scenarios in this study (②), two examples from CSG holding ponds (③④), and other contexts from forest soil (①), landfill (⑤), and wastewater (⑥). ① Methane emissions from forest soil was 0.31 mg/m²/d[1]; ③ Methane emissions from a CSG holding pond in the Surat Basin in Queensland was reported to be~ 8840 mg/m²/d[2]. ④ Methane emissions from a CSG holding pond in NSW was reported to be 150 mg/m²/d in winter and 260 mg/m²/d in summer[3]; ⑤ Methane emissions from an urban sewage treatment plant inlet was 28900 mg/m²/d[3]; ⑥ Methane emissions from an 80 ha landfill site was 35000 mg/m²/d[3].

Such large variations from an order of magnitude (NSW example) to two orders of magnitude (Queensland example) higher than the emissions estimates in the scenarios may indicate that the scenarios represent a significant underestimate of methane emissions and suggest the necessity of an intensive investigation on the emissions from CSG holding ponds in Queensland.

Key knowledge gaps

This study revealed that there are significant knowledge gaps about actual emissions from CSG holding ponds and the key factors that contribute to emissions from such ponds.

- The true emissions potential of CSG holding ponds in Queensland remains unknown. The presence of methane emissions from brine ponds is also unknown. It would seem prudent to

[1] Feng, H., Guo, J., Peng, G., Ma, X., Kneeshaw, D., Chen, H., Liu, Q., Liu, M., Hu, C. and Wang, W. (2023) Global estimates of forest soil methane flux identify a temperate and tropical forest methane sink. Geoderma 429, 116239.

[2] Kelly, B.F.J., Lu, X., Harris, S.J., Neininger, B.G., Hacker, J.M., Schwietzke, S., Fisher, R.E., France, J.L., Nisbet, E.G., Lowry, D., van der Veen, C., Menoud, M. and Röckmann, T. (2022) Atmospheric methane isotopes identity inventory knowledge gaps in the Surat Basin, Australia, coal seam gas and agricultural regions. Atmospheric Chemistry and Physics 22, 15527-15558.

[3] Day, S., Tibbett, A., Sestak, S., Knight, C., Marvig, P., McGarry, S., Weir, S., White, S., Armand, S., van Holst, J., Fry, R., Dell'Amico, M., Halliburton, B. and Azzi, M (2016) Methane and volatile organic compound emissions in New South Wales. CSIRO, Australia

conduct studies that measure methane (and other gas) emissions from these CSG holding ponds to obtain some baseline values.

- There is no information on microbial communities of CSG holding ponds. Microbes play key roles in not only creating methane but also its mitigation through a process called methanotrophy.
- Existing water chemistry data are limited to the water itself and did not measure particulates which have settled to the bottom and are almost certainly the major source of organic carbon in CSG holding ponds.
- The contributions of inorganic carbon such as bicarbonate are unknown but are presumably related to the ability of microbes and algae to use this pool of carbon and mobilise it for biological purposes.

Options for future uncertainty reduction

Based on the data presented here, the primary course of action would be a field campaign to quantify methane emissions and fill the knowledge gaps on CSG holding ponds to provide a better understanding of the contribution holding ponds may make towards the methane budget in Queensland. A more substantive exploration of mitigation options could result from a study that seeks to understand the contribution of microbes as agents that reduce and increase emissions in these ponds.

This review of the literature has provided important experimental design information that may assist in avoiding some of the limitations of other pond emissions studies. Key considerations include:

- Replication. From the literature, natural, and even CSG industry ponds, vary considerably in terms of their water chemistry and their emissions. In any future work it will be important to consider selection of CSG related ponds to include ponds of different types (e.g., produced water ponds and brine ponds in different sizes) and samples collection should include waters derived from both the Surat and Bowen basins to obtain a harmonised suite of comparable data.
- Methods. Measuring emissions in a combination of top-down and bottom-up approaches would be valuable for individual ponds. Further, ensuring both ebullitive and diffusive emission pathways are considered will be important. Suggested bottom-up approaches include the use of floating flux chambers, bubble traps (for ebullitive emissions) and the use of Isoflask sampling to measure dissolved (diffusive) methane. Satellite sensing as a top-down method could be explored.
- Unknowns. Little is known about organic carbon of sediments in the CSG holding ponds and contributions of inorganic carbon, which should be covered in future work. From a microbial perspective, there is little known about the potential of the microbial communities in the ponds to mitigate or contribute towards methane generated in the ponds. The absence of these data represents a troubling knowledge gap.
- Mitigation. The review of the non-CSG related waterbody literature highlights engineering and microbiological approaches that could be used to mitigate emissions from CSG water holding ponds. These should be further explored in subsequent work.

Part I Background

1 Introduction

It has been forecast that global warming may reach 1.5 °C above pre-industrial levels by 2030; and many countries consider any warming beyond 2 °C to have dangerous consequences (IPCC, 2018). Environmental damage from increasing temperatures will likely include marked effects on the natural world and its ecosystems, along with impacts to humanity associated with these changes. Thus, combating the dangerous impacts of global warming is likely one of the most critical challenges humanity faces over the next few decades.

Global warming considered to be largely caused by gases that trap heat in the atmosphere. The contributing gases are called 'greenhouse gases'. Methane is the second most important greenhouse gas after carbon dioxide and the comparative impact of methane is 32 times greater than carbon dioxide over a 100-year period (Holmes et al., 2013). Atmospheric methane (1870 ppb in 2020) has nearly tripled since pre-industrial times (700 ppb) (Neininger et al., 2021; Rosentreter et al., 2021). Abating global warming puts pressure on industries that produce significant quantities of greenhouses gases to reduce their emission profiles. Quantifying and reporting such emissions are becoming more rigorous and specific in the context of global efforts to have net zero emissions. Australia, for example, aims to achieve a 30% reduction in methane by 2030 as part of the Global Methane Pledge (https://minister.dcceew.gov.au/bowen/media-releases/australia-joins-global-methane-pledge). In the energy sector, this means increased efforts to mitigate emissions using new and innovative methods. In the short to medium term, however, gas is likely to be an important, dispatchable source of energy that replaces coal when renewable energy is either unavailable or unsuitable (Boersma and Jordaan, 2017; Gürsan and de Gooyert, 2021).

Given its contribution to global warming, managing methane emission is critical. Methane emissions from the gas industry (particularly onshore) are well characterised for several parts of the industrial process. Previous studies, for example, have focussed on fugitive emissions from gas infrastructure, or gas compression plants (Day et al., 2016a, 2016b, 2015, 2014; Ong et al., 2019, 2017) in Queensland, New South Wales and Northern Territory. Fugitive emissions of methane released from produced water relating to the operation of a facility was accounted by National Greenhouse and Energy Reporting (DISER, 2022). There are, however, gaps in our understanding of emissions across the entire gas industry. One such gap is our understanding of methane emissions biologically generated from water holding ponds associated with coal seam gas (CSG) operations. It is important to note that as it is currently described, methane biologically generated from CSG holding ponds would not technically be a 'fugitive emission' *sensu stricto.* However, community expectations would likely view any emission from a process an industry undertakes as part of their day-to-day operations to be part of their emissions profile. Water, as a co-product of oil and gas operations in the Energy sector, is referred to as produced water, formation water or associated water.

In Queensland, the majority of onshore gas industry is conducted in the Surat and Bowen basins with its primary form as CSG, and other resources in Queensland (for example the Adavale Basin or the Cooper Basin) being both conventional oil and gas systems or related to shales. CSG exploration activity in the Galilee Basin was intermittent during 2008 to 2018 and recently the

Glenaras multi-lateral pilot has been operational since 2018. The Adavale Basin is currently under-explored and considered frontier basin with a confirmed petroleum system. For a more detailed description of the geological settings of these basins, see Apx A and references cited therein. Due to the larger water production resulting from CSG activities, this study is mainly focused on CSG holding ponds in the Surat and Bowen basins, however, where data on water chemistry or emissions for the Cooper, Galilee and Adavale basins are available, these will be explored herein.

Coal seam gas production and the role of water

Coal seams store both gas and water. The water, which is under hydrostatic pressure, holds the gas in place. The removal of water in the coal seams reduces the pressure, enabling the gas to be released (desorbed) from the coal seams and allowing the gas and produced water to be carried to the surface. Hence, the commercial production of naturally occurring CSG in Australia and worldwide often requires the extraction and co-extraction of significant volumes of formation water. For instance, current associated water extraction by the CSG operators in the Surat Basin and Bowen Basin is around 54,000 ML/year (OGIA, 2021).

The quality and quantity of the produced water, however, depends on the geology and hydrogeology of the target formation as well as the underlying and overlying units and will change over the life of a project (Robertson, 2018). Coal is a naturally fractured rock with water largely occupying the fractures. The frequency and number of fractures within the coal result in more permeable coals. This permeability is an action of the composition of coal stress regime and subsequent geological events, but in essence, higher permeability allows easier water and gas flow through the coal seam. It is important to note that this groundwater has often been in the subsurface for a very long period and may uptake many solutes. These solutes can be dissolved into the water from the coal itself or may come from the water interacting with other rock strata as it makes its way into the coal seam. Hence the quality of produced water varies across regions, but is typically high in pH, total dissolved solids and bicarbonate (APPEA, 2018; Dahm et al., 2011; Fell, 2014; Plumlee et al., 2014; Robertson, 2018; Taulis and Milke, 2007).

When CSG is extracted via a well, produced water, co-produced with the gas, is separated in a phase separator, a device (Figure 2) that allows water to be split from the gases that are being produced. After separation, the produced water is moved to a water management pond through an array of supporting infrastructure such as pipe networks and pumps.

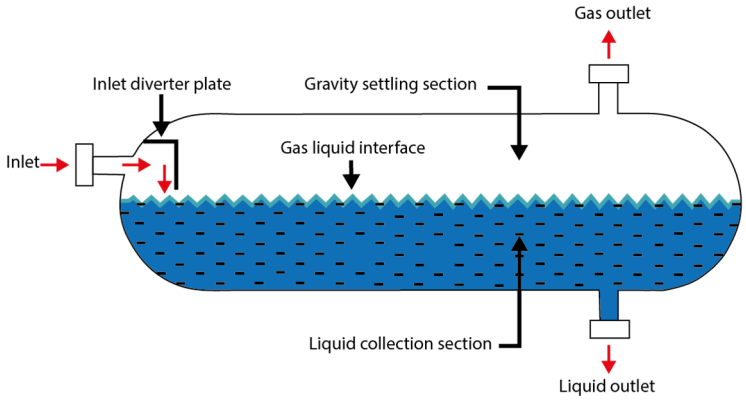


Figure 2 A schematic diagram of a separator used for CSG wells

Since produced water (or raw water) from coal seams is often 'brackish', its direct use is limited (Nghiem et al., 2011; Plumlee et al., 2014; Robertson, 2018). The produced water in the water management pond requires treatment before turning into a water resource for use within the community. The CSG produced water treatment process is shown in Figure 3. A **produced water pond** (or called raw water pond, water management pond and feed pond depending on different companies) is primarily used for water storage prior to the processing of the water in the treatment plant. Treatment normally involves solid removal and desalination (Millar et al., 2016; Nghiem et al., 2015, 2011). Produced water from water management ponds often contain large particles or foreign material such as soil, sediment, algae or other materials, hence these solids in the management ponds will be removed first before going to the next treatment step. Desalination is used to remove total dissolved solids (e.g., calcium and magnesium) from the produced water by ion exchange and reverse osmosis (RO). A **treated water pond** is normally located downstream of the desalination plant. Treated water can be used for various beneficial uses such as for agriculture purposes or recreational use (Robertson, 2018). A **brine pond** normally receives the RO rejected water. Other types of ponds also exist where water from non-coal aquifers is being used for various purposes (e.g., dust mitigation or to create liquids for various subsurface operational processes). However, non-coal associated groundwater, is outside the scope of the current study and is probably the least significant of these water sources in terms of volume.

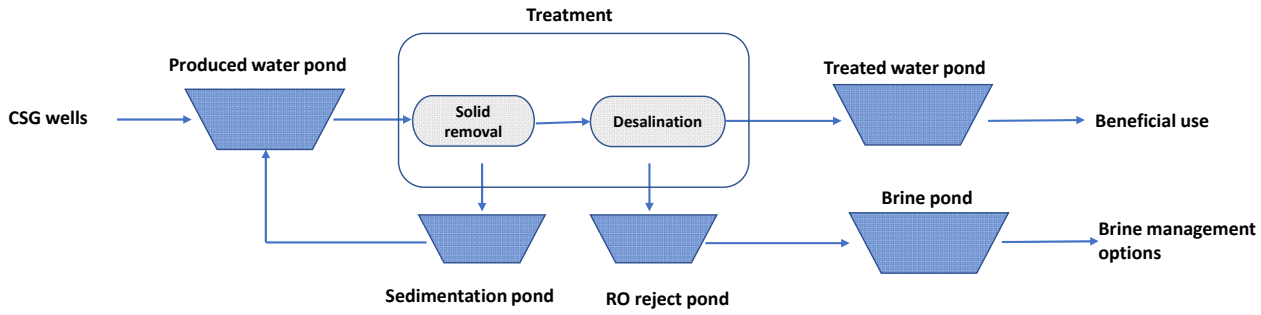


Figure 3 A schematic diagram of a CSG water treatment process

While significant research has previously been conducted to investigate the integrity of such holding ponds and their capacity to not harm the public or the local environment (APPEA, 2017; Healy et al., 2011; Navi et al., 2015; Young, 2005), very limited work has been recorded in the public domain on the potential of water holding ponds as sources of methane, carbon dioxide or other greenhouse gases into the atmosphere. While methane from the CSG holding ponds could be derived from geological methane being carried over in water from the separator, it could also be generated by methanogenic archaea utilising organic materials in the formation water collected. This may also include biologically fixed inorganic carbon from bicarbonates (fixed through algal or microbial processes) (Gardner et al., 2013; Yanyou Wu, 2021).

Preliminary CSG holding pond methane data in the literature

At the outset of the study only limited data had been identified in the open-file literature on methane emissions of CSG holding ponds and their contribution to the global carbon cycle, which included a few ground-based or air-based methane measurements in Queensland and New South Wales.

An elevated methane plume measured by CRDS was detected within 50 m distance from a CSG holding pond in the downwind direction having a peak methane concentration of 2.107 ppm which is 0.333 ppm higher than the background level (Iverach et al., 2015).

Elevated methane concentrations were also detected 1 km away from another raw water pond related to CSG activity in both 2018 and 2019 with excess methane concentrations over background being 0.2 ppm and 1.5 ppm, respectively (Lu et al., 2021).

Methane emissions of 221kg/h (~8840 mg/m^2/d) from a raw water pond related to CSG in Queensland was reported by Kelly et al. (2022).

An example from a CSG holding pond in NSW showed the mean methane emissions were from 150 mg/m^2/d in winter to 260 mg/m^2/d in summer (Day et al., 2016b).

Such huge variations of methane emission measurements or estimates based on a few CSG holding ponds from different researchers indicates the necessity of an intensive investigation on the emissions from CSG holding ponds in Queensland.

Natural analogues for CSG holding ponds

Numerous methane emissions studies from natural water bodies may provide some insights into the potential contributions of the CSG holding ponds to methane emissions. In brief, the literature review has demonstrated that freshwater bodies (e.g., lakes, ponds, reservoirs, rivers, impoundments) are important contributors to global methane emissions (Bastviken et al., 2011; Kirschke et al., 2013; Saunois et al., 2020). Natural lakes contribute 70% of the freshwater methane emissions (Sanches et al., 2019). Recent global methane emission estimates from a range of freshwater bodies between 2000 and 2009 amount to 159 teragrams (Tg) of methane per year (Saunois et al., 2020). Despite a range of landmark studies into methane emission from freshwater bodies significant knowledge gaps remain. Any comparison of CSG holding ponds with alternative freshwater bodies is also hampered by a lack of data from a range of climatic zones. It should be emphasised that currently, most studies are concentrated in boreal and temperate environments and different scenarios are expected to occur in, for example, subtropical or tropical areas (Mendoza-Pascual et al., 2021). Furthermore, CSG produced water and brine contain significantly higher pools of inorganic carbon than most natural water bodies, and part of this project will explore the potential for this otherwise immobile carbon store to act as a potential reservoir for emissions (Gardner et al., 2013; Ghafari et al., 2009; Mokashi et al., 2016; Y. Wu, 2021; Yanyou Wu, 2021).

The aim of this study is to:

1) Review the current literature on methane emissions from natural and anthropogenically built waterbodies such as lakes, ponds or reservoirs, to provide context on relative contributions, and harmonise any data identified for comparative purposes.

2) Collate water chemistry and microbiological data, as well as methane emissions records related to CSG holding ponds from industry and publicly available datasets.

3) Develop emission monitoring strategies for CSG holding ponds such as sampling protocols, operational factors, water chemistry analysis, and microbiological analyses, to ultimately provide

sampling workflows to assess contributions from water storage and handling treatments at CSG sites.

4) Examine potential mitigation strategies for methane emissions reduction from CSG water holding ponds.

2 Existing knowledge on emissions and waterbodies

Little is known about the potential for methane (or other gas) emissions from CSG holding ponds. It is possible to gain some understanding of this potential through exploration of data from natural waterbodies (lakes in particular) or anthropogenically constructed waterbodies such as ponds or dams in urban settings. Of these environments, lakes and urban ponds are useful models for emissions potential for CSG holding ponds as they are more static environments with a greater degree of stratification (Boehrer and Schultze, 2008).

Inland waters are a major source of methane globally and anthropogenic construction of reservoirs and urban ponds are thought to enhance this flux (Bastviken et al., 2004). Small waterbodies are becoming increasingly recognised as hotspots for methane emissions (Audet et al., 2020; Grinham et al., 2018; Holgerson and Raymond, 2016). For examples, small water bodies (<0.1 ha) were reported emitting 40% of all diffusive methane emissions, even though they compose only about 9% of the total area of lentic fresh waters (Holgerson and Raymond, 2016).

Methane or other emissions in CSG holding ponds are likely to be controlled by organic matter/carbon that is introduced during production, or results from the degradation or alteration by microbes of that organic matter. However, there may also be a role for inorganic carbon as a pool of carbon that can be altered. This is in addition to the introduction of other organic matter from around terrestrial water bodies.

2.1 Source of organic matter in water bodies

Organic matters in lakes are directly sourced from 1) autochthonous inputs from photosynthetic organisms (phytoplankton, benthic algae, and/or plants on the shore of the water body) that occur in or near the lake, or 2) allochthonous inputs from vegetation/algae etc found in the lake catchment area (Cerling et al., 1997; Schenk et al., 2021). Obviously organic matter can be in the form of moribund (dead) organisms that consume these primary producers.

Several factors influence the rate of methanogenesis from these environments such as type of organic matter in certain environments. In sediments sourced from Diamond Lake in southwest Michigan (USA), methanogenic communities responded more (e.g., produced more methane) in response to algal biomass than terrestrial biomass (William E. West et al., 2012).

2.2 Degradation of organic matter in water bodies

In freshwater waterbodies, methane is primarily produced in anoxic sediments by methanogenic archaea during anaerobic metabolism (Figure 4). Methane represents the last step of organic matter decomposition in anaerobic water or sediment layers (Kuivila et al., 1989; Lovley and Klug, 1986). In brief, this process begins as organic material, for example, a leaf or a dead animal (F in Figure 4), enters a waterbody and components within the leaf (sugars, proteins, and nucleic acids) are very rapidly degraded by aerobic microbes (A in Figure 4) within the water body. While some of this material makes its way to methane through the death of these organisms; only the more

recalcitrant organic material (for example lignin, or waxes from the leaf cuticle) makes it to the sediment where these components are available for further degradation through heterotrophic anaerobic microbes (B in Figure 4). Within the sediment, these components are sequentially degraded by syntrophic partners of methanogens (C in Figure 4) through increasingly simple forms[4] until the carbon is in the form of methane by methanogens (D in Figure 4). This methane frequently accumulates as bubbles trapped within the sediment. Often, as these bubbles increase in size, they overcome the resistance the sediment provides and they move to the ponds surface, exiting the process as an ebullitive emission. During this process, methane could be oxidised and consumed by methanotrophs (methane eating microbes, E in Figure 4).

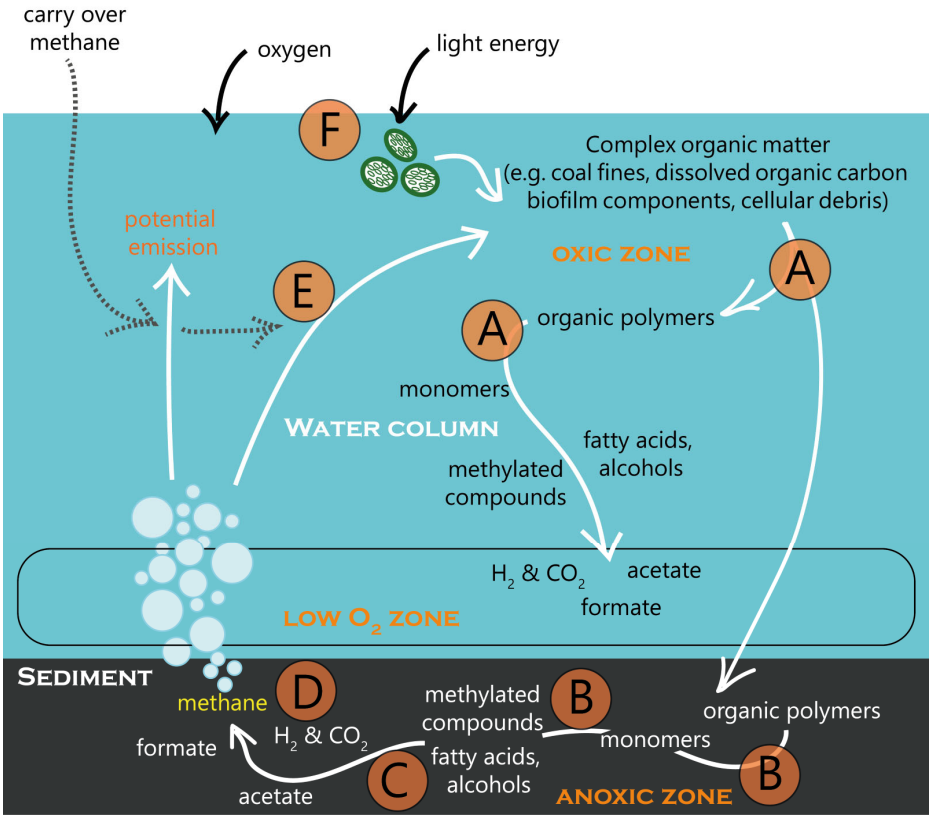


Figure 4 Conceptual map of possible abiotic and biotic processes within a waterbody. (A) heterotrophic aerobic microbes (mostly bacteria); (B) heterotrophic anaerobic microbes (exclusively bacteria and archaea); (C) syntrophic partners of methanogens and other taxa (frequently clostridia or proteobacterial species; (D) methanogens (see Figure 5 for a detailed explanation of the multiple types of methanogenesis); (E) methanotrophs (see Table 1) and (F) photosynthetic autotrophs including algae and Cyanobacteria.

2.3 Organisms that generate methane - methanogens

The process of making methane biologically is called methanogenesis and is undertaken by a group of mostly related microbes within Euryarchaeota (one of the better studied phyla[5]) in the

[4] This is a simplification. In reality, carbon contained within compounds can become more complex through incorporation into biomass, however, in net terms, a significant proportion of the carbon in anoxic sediments becomes methane over time.

[5] A phylum (plural phyla) is a high level i.e., broadly encompassing group of organisms. For example, all vertebrates (fish, mammals, amphibians etc) are part of phylum Chordata.

archaeal domain (one of the three domains[6] of life, see Figure 5) called methanogens. For completeness it should be noted that some non-euryarchaeotal methanogens do occur, notably members of the recently described phylum Bathyarchaeota (Evans et al., 2015).

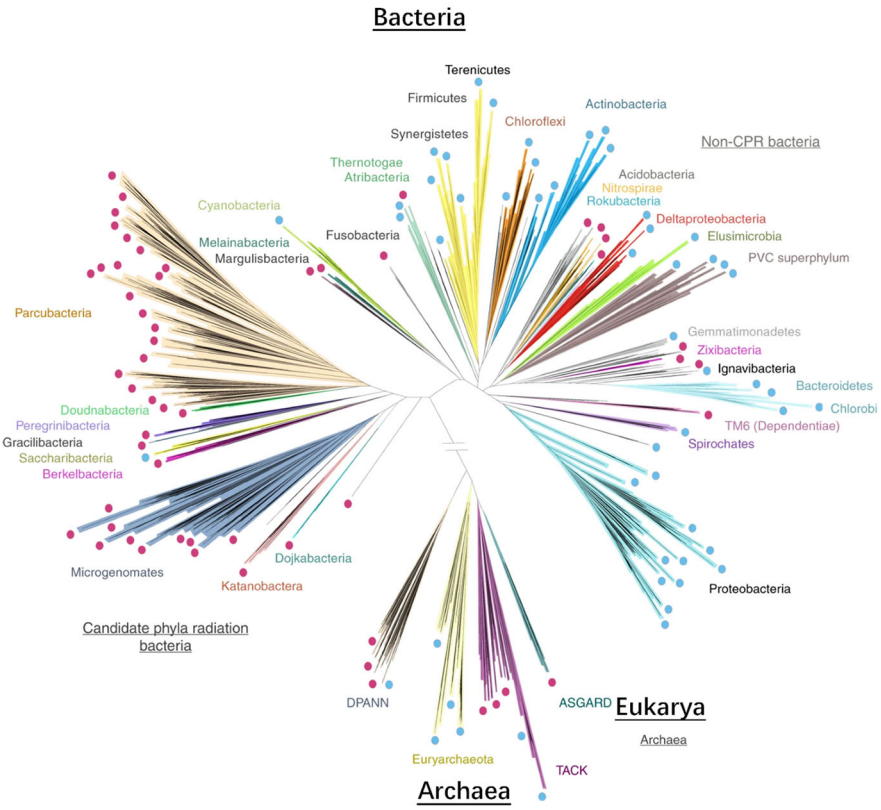


Figure 5 Phylogenetic tree of life constructed using genetic information. Organisms/phyla that are closer together and more closely related. At the bottom-centre of the tree, it is the Archaea phyla which include TACK6, Euryarchaeota and DPANN. Most methanogens are members of the archaeal phylum Euryrchaeota.

Image and underpinning analysis by Hug et al. (2016)

Methanogens are obligate anaerobes[7] that thrive under anoxic conditions and often, but not exclusively, exist in nutrient-poor settings where all electron acceptors other than carbon dioxide have been depleted (Atlas and Bartha, 1998; Knoll et al., 2012; Slonczewski and Foster, 2017). Methanogens are generally divided into three groups based on their physiology (i.e., the materials they use for generating methane). **Hydrogenotrophic methanogens** mostly oxidise hydrogen and reduce carbon dioxide for methanogenesis with exceptions of some methanogens in this group using formate, instead of hydrogen for the reduction of carbon dioxide to methane (Kurth et al., 2020). **Methylotrophic methanogens** can make methane from various compounds with methyl groups such as methanol or various methylated amines. **Acetoclastic methanogens** can convert acetate to methane.

It is important to note that this represents a very high-level summary, there is considerable diversity among methanogens and some have more complex or nuanced physiology than is

[6] There are three groups of life on earth: Bacteria, Archaea and Eukarya. Both bacteria and archaea are single celled, though they are not closely related at all. The Eukarya includes life as most people would recognise it: plants, fungi and animals along with a host of other simple eukaryotes.

[7] Obligate anaerobe means that these microbes can only survive in the absence of oxygen.

described here. For further information, readers should consider some of the modern reviews on the group (Costa & Leigh, 2014; Kurth et al., 2020 for material on diversity of chemicals used by methanogens, Buan, 2018 or Bueno de Mesquita et al., 2023 for information on the energy metabolism of methanogens).

Finally, it is important to note that while most methanogens are not extreme halophiles and probably cannot survive in brines, there are groups of methanogens that specialise in this environment. For example, there are numerous examples of halophilic (salt-loving) methanogens from the genus Methanohalophilus within the euryarchaeal phylum. All members of this genus are methylotrophic. They include Methanohalophilus euhalobius (Obraztsova et al., 1987), Methanohalophilus halophilus (Zhilina, 1986), Methanohalophilus levihalophilus (Katayama et al., 2014), Methanohalophilus mahii (Paterek and Smith, 1988) and Methanohalophilus portucalensis (Boone et al., 1993). In 2017, a new lineage of salt loving methanogens was also identified and placed in a distantly related lineage called the Methanonatronarchaeia (Sorokin et al., 2017). Since quite a number of CSG holding ponds are brine ponds, the knowledge of salt-loving methanogens is critical for investigating methane emissions from brine ponds.

2.4 Organisms that consume methane - methanotrophs

The amount of methane in a freshwater body is also controlled by a counteracting microbial process called methantrophy (literally nourished by methane) (Conrad, 2009; Fuchs et al., 2016). A bacterial process (though some archaea also engage in this process) in which methanotrophs oxidize methane to CO_2 under oxic and anoxic conditions using a range of diverse electron acceptors, and play an important role in the carbon cycle (Guerrero-Cruz et al., 2021; Knoll et al., 2012). These organisms inhabit a wide range of ecosystems including a range of marine and freshwater bodies, forests, grasslands, and deserts. Methanotrophs are a group of microorganisms consisting of members from both the bacterial and archaeal domains (Table 1).

Most of the well-studied methanotrophs are from the phylum Proteobacteria (Figure 5), which include members of the gammaproteobacterial class and the alphaproteobacterial class (Stein et al., 2012, Table 1). Under aerobic conditions, proteobacterial methanotrophic bacteria consume methane and oxygen to produce formaldehyde, a precursor molecule that becomes incorporated into one of two metabolic pathways that eventually results in the formation of CO_2. One type of metabolic pathway, known as the ribulose monophosphate (RuMP) pathway, is associated with the Gammaproteobacteria (families Methylococcaceae and Methylothermaceae). The other type of metabolic pathway uses the serine pathway of carbon assimilation and is employed by the Alphaproteobacteria (families Methylocystaceae and Beijerinckiaceae) (Stein et al., 2012). The phylum Verrucomicrobia (family Methylacidiphilaceae, Table 1) also aerobically oxidise methane (Op den Camp et al., 2009). Aerobic methane oxidation has also been shown to occur in some cases in anoxic environments whereby internally (i.e., intracellularly) produced oxygen is used to oxidise the methane. This ability is expressed by a member of the phylum “Methylomirabilota” in Table 1 (Ettwig et al., 2010). Members of this group are poorly understood as they are yet uncultured.

While various methanotrophs exist in anoxic environments, they are typically not highly abundant or they live only in quite specialised environments, such as in marine and lake sediments, rice paddies and soils. In these environments, methanotrophy occurs whereby compounds other than oxygen (e.g., nitrate, nitrite, iron, manganese, or sulphate) are used as electron acceptors. These groups of microorganisms belong to the archaeal domain and work in association (or consortia)

with sulphate-reducing bacteria to oxidise methane. Archaeal taxa capable of this metabolism belong to the Methanosarcinales order in Table 1 (Offre et al., 2013). The exact details of their metabolism involving methane is still a subject of debate.

Regardless of the type of methanotrophy, this process has been estimated to reduce methane emission to the atmosphere from between 30 to 99% (Bastviken et al., 2008; Frenzel et al., 1990). Globally it is a key process for mitigation of methane emissions from anoxic environments.

Table 1 Taxonomy of all recognized methanotrophs according to their phylogenetic classification. Taxa from both the Bacterial and Archaeal domains are listed. Information adapted from Guerrero-Cruz et al. (2021).

Domain	BACTERIA					
Phylum	Proteobacteria (synonym Pseudomonadota)				Verrucomicrobiota	"Methylomirabilota"
Class	Gammaproteobacteria		Alphaproteobacteria		"Methylacidiphilae"	"Methylomirabilacaea"
Order	Methylococcales		Hyphomicrobiales		"Methylacidiphilales"	"Methylomirabilales"
Family	Methylococcaceae	Methythermaceae	Methylocystaceae	Beijerinckiaceae	"Methylacidiphilaceae"	"Methylomirabilaceae"
Genus	*Methylococcus*	*Methylothermus*	*Methylosinus*	*Methylocella*	*Methylacidiphilum*	*"Methylomirabilis"*
	Methylomonas	*Methylohalobius*	*Methylocystis*	*Methylocaspa*	*Methylacidimicrobium*	
	Methylobacter	*Methylomarinovum*		*Methyloferula*		
	Methylomicrobium					
	Methylosarcina					
	Methylocaldum					
	Methylogea					
	Methylosoma					
	Methyloparacoccus					
	Methyloglobulus					
	Methyloprofundus					
	Methylomarinum					
	Methylovolum					
	Methylomagnum					
	Methylosphaera					

Domain	ARCHAEA			
Phylum	Euryarchaeota			
Class	Methanomicrobia			
Order	Methanosarcinales			
Clade		ANME-1	ANME-2	ANME-3
Family	Methanoperedenaceae			
Genus	*Methanoperedens*			

2.5 Pathways of methane emissions from waterbodies

Ebullition, diffusion and plant-mediated transportation are three major pathways of methane emission in natural or anthropogenically built waterbodies (Figure 6, Bastviken et al., 2004; Grinham et al., 2018). Ebullition is via ebullitive (bubbling) emissions where methane overcomes hydrostatic pressure and rise to the surface in forms of bubbles. Diffusion is the process by which methane may escape from ponds, and this is essentially dissolved methane being released at the water surface, termed as diffusive emissions. Finally, in natural ponds, but not in CSG holding ponds, plants may mediate transport of methane out of ponds and waterlogged soils.

Of these processes, ebullition (bubbling) is often the dominant form of methane emission from shallow waterbodies but the least understood (Bastviken et al., 2011; Burke et al., 2019; Coulthard et al., 2013; Grinham et al., 2018; Joyce and Jewell, 2003). Depending on water chemistry in various environments, ebullitive methane emissions account for 50 to 95% of total methane fluxes (Bastviken et al., 2010; Malyan et al., 2022; Natchimuthu et al., 2014; Peacock et al., 2019; Walter et al., 2007). The uncertainty in ebullition rates is due to highly episodic releases, triggered by environmental conditions such as changes in water level and atmospheric pressure (Goodrich et al., 2011; Weyhenmeyer, 1999), wind (Keller and Stallard, 1994) or no apparent trigger (Joyce and Jewell, 2003), in combination with large spatial variations (Laurion et al., 2010; Wik et al., 2013), as well as lack of measurements able to account for these heterogeneities (Wik, 2016). Even though

ebullition is difficult to predict, Wik (2016) was able to identify two strong predictors of ebullition with one being a high methanogenesis rate and the other being a pond depth of less than 6 m. This is because in shallow ponds, bubbles have relatively limited residence times in the ponds as they quickly rise to the surface (Bastviken et al., 2004; Walter et al., 2007). Ebullitive methane emissions from shallow ponds in North America were shown to be as high as 640 mg/m^2/d (Baron et al., 2022).

Diffusively-emitted methane generally accounts for 5 to 37% of the total methane emissions, and it is controlled by both abiotic and biotic factors (Baron et al., 2022; Bastviken et al., 2004; Grinham et al., 2018; Malyan et al., 2022; Peacock et al., 2019; Walter et al., 2007). In other settings, however, diffusive emissions can be the dominant form of emissions for other water bodies. For example, in a study of lowland headwater streams, diffusive emissions accounted for more than 75%, and up to 100% of emissions (Robison et al., 2022).

Plant-mediated transport can be important in some environments. This in part as deep-rooted plants can assist methane in bypassing methanotrophy (using their roots as conduits) in the shallower parts of the soil (Korrensalo et al., 2022).This pathway is outside the scope of this study, as CSG holding ponds typically contain no plants.

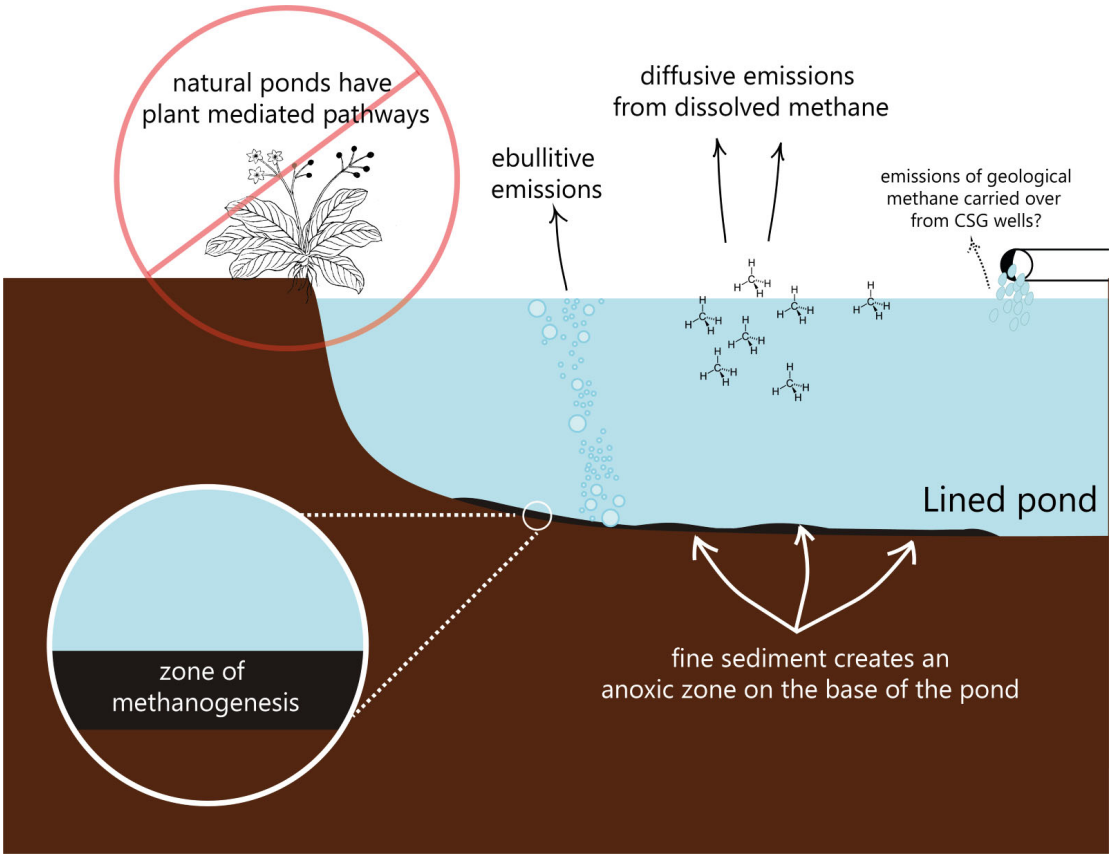


Figure 6 Major pathways of methane emissions from CSG holding ponds include ebullitive and diffusive emissions but not plant-mediated pathways.

3 Measuring methane emissions from waterbodies

There are multiple methods in the literature for measuring atmospheric and dissolved methane, which have been used to determine ebullitive and diffusive components of methane flux from waterbodies. These range from direct sampling of gas and water for later laboratory analysis, to *in situ* measurement of methane in water and air by field instruments (both fixed and mounted on mobile sampling platforms).

Multiple measurements over different areas (such as transects of a lake) and different times (over day night cycles, or seasons) give insight into the variability of methane flux within an area.

These point measurements can also be used to feed statistical models that, when coupled with intensive meteorological measurements, or stable isotope ratio measurements, can derive estimates for methane flux over large areas or from different sources.

Each of these methods for determining methane flux have limitations and advantages, from sample collection or data processing intensity to differences in cost and ease of deployment.

3.1 Sampling

3.1.1 Gas sampling

In order to carry out any laboratory-based analysis of methane, the gas must be firstly sampled into an appropriate container for storage and transport to the laboratory. The type of container used is determined by the type of sample needed, the suitability of use, the holding time between sampling and analysis, and the component of interest. Gas samples may be 'grab sampled' or injected directly into gas tight vials, purpose-made containers such as isotubes or bags (Nauer et al., 2021; Van Holst et al., 2010), drawn in a controlled flow into a container over a long period as a 'whole air sample' (RESTEK, 2017; Zielinska and Fujita, 1994), pumped to collect a large sample from a low pressure source (Sestak et al., 2017), or collected using an aircraft (Kelly et al., 2022). Sampling into bags is potentially more straightforward and practical in the field than sampling into cylinders, however, these samples may not be suitable for longer term storage without loss of the usefulness of the sample.

3.1.2 Water sampling for dissolved methane analysis

Conventionally, water samples at surface and depth for dissolved gases are collected in gas tight sampling devices which can be manually or often remotely activated once the container is at the depth of interest. These range from syringes (Audet et al., 2020), Van Dorn samplers (McClure et al., 2018), glass vials ((Pearce et al., 2023)), isoflasks (Pearce et al., 2023) or other custom assemblies that allow the sample to be collected without any contamination from atmospheric air (Bastviken et al., 2004). Bactericide should be added to these containers to prevent biological activities between the sampling and analysis (Bastviken et al., 2004; Malyan et al., 2022) and the samples containers should be kept cold while in transit to the laboratory (Hounshell et al., 2021).

To measure the dissolved methane in the sample, a volume of the water of the sample is exchanged with an inert gas for the methane to diffuse into; this is known as 'headspace gas analysis' (Snow and Slack, 2002). The headspace gas analysis can be carried out while in the field, or using a portable instrument (Audet et al., 2020; Banks et al., 2019; Bevelhimer et al., 2016). Once the headspace is created in the sample with an inert gas (usually nitrogen or helium), the sample is then mixed thoroughly, so that the methane gas is at an equilibrium within the water and headspace. This headspace gas containing the extracted methane is then withdrawn using a gas tight needle and analysed by gas chromatography. Water temperature and atmospheric pressure at the time of sampling is used to accurately back-calculate the concentration of methane in the water.

Iverach et al. 2020 used a Picarro CRDS to measure groundwater and surface water methane levels in the field by stripping the gas out of solution using a dissolved gas extraction unit and collecting the extracted gas in FlexFoil bags for analysis. Briefly, water is pumped into a hydrophobic membrane, with a counter-flow of gas on the outside of the membrane, known as a sweep gas. Dissolved gas diffuses from the water through the membrane and into the sweep gas stream, where it can be collected and analysed.

3.1.3 Field Instrumentation

More recently, portable devices for measuring single components such as methane or carbon dioxide, using laser based optical absorption methods have emerged and seen extremely wide use; they have been adapted for both dissolved and atmospheric methane. These devices are based on laser-based absorption spectroscopy methods such as tunable laser diode absorption spectroscopy (TLDAS) (Milton, 2005), and forms of cavity-enhanced absorption spectroscopy (CEAS), including CRDS and OA-ICOS (Orr and He, 2014; Shao et al., 2022). These instruments continuously measure concentration of one or more gas components simultaneously (often methane and carbon dioxide) by measuring their absorbance of laser light at specific wavelengths, and may be 'open-path', where the instrument interrogates an air gap, or 'cavity-based', where the air sample is actively, continuously pumped into a chamber, in which laser light is reflected. In both cases the instruments may be fixed or mobile. These instruments are portable and robust, and have been fitted to various platforms to make long term, wide scale gas measurements, including trolleys (Myers et al., 2020), ships (Judd, 2015; Paris et al., 2021), helicopters and cars (Day et al., 2014; Ong et al., 2019), and even smaller vehicles such as UAVs (Gålfalk et al., 2021) and ASVs (Grinham et al., 2011). These versatile instruments may provide a mobile, flexible option for carrying out methane measurements at multiple sites, where long term, fixed field installations are not feasible (Day et al., 2016b).

3.2 Methods for methane emissions flux from waterbodies

In general, measurement for the methane emissions from waterbodies is achieved through a variety of methods to capture ebullitive, diffusive or combined (diffusive plus ebullitive) emissions.

3.2.1 Combined emissions

The floating chamber method is widely used for the greenhouse gas measurement from waterbodies (Bastviken et al., 2004; Erkkilä et al., 2018; Grinham et al., 2018, 2011; Herrero Ortega et al., 2019; Natchimuthu et al., 2014; Peacock et al., 2019; Peacock et al., 2021; Schrier-Uijl et al., 2011). Floating chambers (Figure 7) or similar have a confined airspace above the water surface, and are commonly used in a series suspended across the water body at different intervals to form a transect, or in groups at different point locations on the water body, or combinations of the above to cover the spatial variation (Bastviken et al., 2010, 2004; Chen et al., 2011; Grinham et al., 2018; Herrero Ortega et al., 2019; Natchimuthu et al., 2014). This method is considered inexpensive and straightforward, without intensive data-processing if using manual sampling (withdrawing the collected gas samples from the chamber using a syringe) (Bastviken et al., 2004; Grinham et al., 2011) but does requires ready access to the laboratory for sample analysis (Natchimuthu et al., 2014).

Self-contained automated systems have been designed to sample gases (Figure 7), in which the chamber is vented, allowed to collect, and subsampled into vials for later analysis (Duc et al., 2013). In later advancements these have been fitted with multiple sensors for determining both ebullitive and diffusive flux remotely using sensors (Thanh Duc et al., 2020). Recently, chamber measurements have been made in-situ without the need for subsampling, by connecting the chambers to a portable gas analyser (Day et al., 2013; van Bergen et al., 2019). It is important to note that floating chambers capture both ebullitive and diffusive emissions unless additional calculation using the distribution and variance in the apparent piston velocities or other measurements like dissolved methane concentration are made to account for the different contributions of both. The floating chamber method is good for detecting spatial variation but has its limitations regarding temporal data coverage since there is clear difference between daytime and night time methane flux and also it is challenging for measurements in windy weather conditions (Erkkilä et al., 2018).

Figure 7 Example of an automated floating chamber (Reprinted with permission from Duc et al 2013, Copyright ©(2013), American Geophysical Union)

3.2.2 Ebullitive emissions

Ebullitive methane emissions are obtained by direct capture or measurement of gas bubbles evolving from the surface of a water body, and the deconvolution of any diffusive component. Various approaches and sampling platforms have been designed for this, ranging from manual, labour intensive sampling approaches, to automated samplers and in-situ measurement options.

The most commonly used, and straightforward ebullitive emission measurement is trapping bubbles using inverted funnels. Ebullitive methane (methane bubbles) from a waterbody can be measured through inverted funnels coupled to gas collector initially filled with water. Partially submerged funnels (moored or floating), with water filled reservoirs or tubing have been used as bubble traps (Figure 8) and all the pre-existing air needs to be removed before the commencement of sampling. Rising bubbles ascend into the funnel and displace the water in the gas collector, providing a measurement of volume over the deployment or 'incubation' time and methane may be withdrawn from the gas collector through a septum by a gas-tight syringe or through other fittings to be collected into isotubes or bags for analysis (Burke et al., 2019; Keller and Stallard, 1994; Rosa et al., 2003; van Bergen et al., 2019; Wik et al., 2013). The sampling duration using funnels should be less than 20 hours to avoid diffusive emissions being captured (Malyan et al., 2022).

Surveys of ebullitive flux have also been performed using an autonomous surface vehicle (ASV), fitted with an open path infrared beam detector interrogating methane concentrations at the water surface (Grinham et al., 2011). This AVS can operate continuously throughout day and night under all weather conditions without human intervention to provides repeatable measurement trajectories with the aim to overcome potential underestimates of methane due to point measurements using floating chambers alone. Results showed that an estimated 97% of the total methane release was highly localised to 1.8-7% of the overall surface area of Little Nerang Dam (Grinham et al., 2011). The authors suggested that this highlighted the need to monitor significant regions or, ''if not the entire, water storage" to obtain accurate flux measurements.

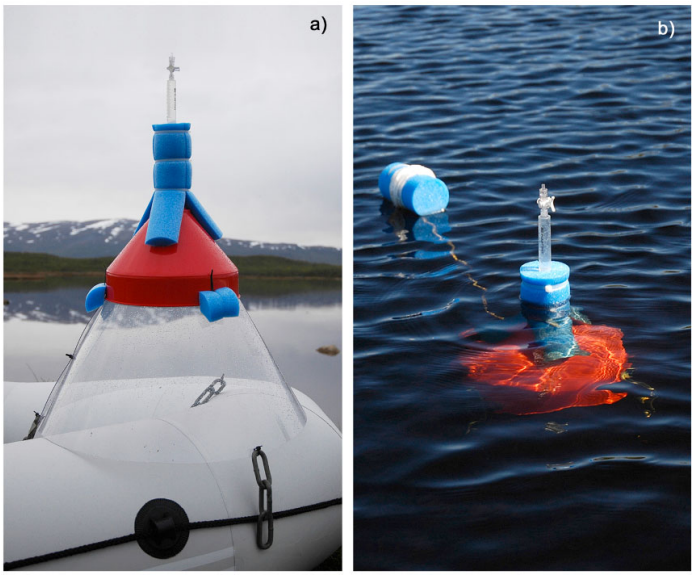


Figure 8 Example of a funnel bubble trap (Reprinted with permission from Wik et al 2013. Copyright (2013) American Chemical Society)

3.2.3 Diffusive emissions

Commonly, the diffusive component of the emissions is determined by measuring the dissolved methane concentration of the surface waters (details in section 3.1.2) and comparing with the maximum solubility of methane under conditions of temperature and salinity. Once this information is known, it is possible to estimate the component of this dissolved methane that will be released to atmosphere, in the conditions of atmospheric concentration, temperature, wind speed and turbulence (often collectively described as the 'piston velocity') (Bastviken et al., 2004).

As mentioned previously, it is possible to estimate the proportion of diffusive methane using results from floating chamber measurements by comparing the difference in predicted flux (based on water concentration), and the actual concentration measured in the chamber. Alternately, an engineering approach to isolate the diffusive component can be taken; in one example a plastic 'bubble shield', larger than the chamber footprint, was fitted underneath a floating chamber and secured using metal wire. This was intended to deflect the bubbles from the chamber, so the chamber only measured diffusive flux, and was shown to be effective with fairly consistent results for diffusive flux obtained across multiple chambers (Bastviken et al., 2010; Wik, 2016).

3.3 Atmospheric methane

The use of UAVs (unmanned aerial vehicles) for measurement of atmospheric methane has been limited by the availability of light payloads for methane monitoring with sufficient accuracy. However, there are now multiple examples of small-scale UAV surveys using light, fast-response infra-red instruments or low cost semiconductor-based gas sensors (Bel Hadj Ali et al., 2020; Gålfalk et al., 2021) to examine methane flux from smaller area sources such as landfills.

Direct eddy covariance (EC) emissions measurement has grown in popularity for long-term monitoring of terrestrial and lake-dominated landscapes (Burba, 2021; Deemer et al., 2016; Erkkilä et al., 2018; Huotari et al., 2011; Vesala et al., 2014). EC is a technique which calculates greenhouse gas flux based on mean air density and instantaneous deviation in vertical wind speed and gas concentrations (Deemer et al., 2016). EC is not designed to account for any small-scale spatial variability from different types of areas that lie within the footprint of the measurement (Thanh Duc et al., 2020), instead, it can provide gas flux estimates over a much larger source area (Aubinet et al., 2012). EC measurements are, however, not only quite expensive in terms of equipment and but also require extensive post-processing of data and also have poor sensor performance during wet conditions (Deemer et al., 2016; Peltola et al., 2013; Vesala et al., 2014).

At an even larger scale, Aircraft sampling of atmospheric air (methane) has been used in many airborne methane emissions and isotope studies to estimate and identify methane sources on the ground (Fisher et al., 2017; Kelly et al., 2022; Menoud et al., 2022; Neininger et al., 2021). A study by Neininger et al 2021 used such methods to examine differences between airborne measurements (Top down 'TD') of methane in Queensland CSG areas, and inventories estimated from local activities and industries (bottom up 'BU') estimates, including from a CSG brine pond in the study area.

Satellite sensing is one of the approaches that can provide global empirical methane emissions measurements by retrieving atmospheric methane column concentrations with unit sensitivity down to the surface through measuring spectrally resolved backscattered solar radiation in the

shortwave infrared (Jacob et al., 2016). It has played an important role in detecting large emission events like extreme methane leakage from a natural gas well blowout and estimating emissions on global and regional scales (de Gouw et al., 2020; Gao and Vollrath, 2023; Guanter et al., 2021; Jacob et al., 2022, 2016; Ong et al., 2017; Pandey et al., 2019; Sadavarte et al., 2021; Varon et al., 2021; Wecht et al., 2014). One of the satellite-borne remote sensing instruments launched by GHGSat Inc was designed to measure greenhouse gases, having an effective spatial resolution of 50 x 50 m^2 over targeted 12 x 12 km^2 scenes (Jervis et al., 2021; Ong et al., 2017). It captures spectral features in the electromagnetic spectrum specific to methane and carbon dioxide using passive optical remote sensing. This method has been attempted to monitor the emissions of CSG sites in NSW (Ong et al., 2017), showing the presence of an emissions plume in the study area. Using satellite sensing to estimate methane emissions is limited by the detection limits of different imagery sensors, ranging from 2400 to 240000 kg/day (Jacob et al., 2022; Ong et al., 2017). Some methane sources over water are also intrinsically difficult to observe from space such as wastewater facilities, agricultural reservoirs and estuaries (Jacob et al., 2022).

Part II Data collation and analysis

4 Natural and anthropogenically built waterbody data sourced from the literature

4.1 Data collation

Methane emission data were collated from 59 peer-reviewed journal articles published between 1977 and 2022. Along with emissions data, data were also collected relating to waterbody area and depth, temperature, climate, pH, dissolved organic carbon, total phosphorous (TP), total nitrogen (TN) and location of the waterbodies where they are available (Apx Tables C.1, C.2 and C.3) The types of waterbodies examined included urban ponds, natural lakes, reservoirs from different arctic, temperate, subtropical and tropical climates across different continents including Asia, North America, South America, Europe and Oceania.

4.2 Data availability

This study focuses on publicly available data; hence the most important resources are validated/peer reviewed papers in journal publications. Availability of methane emission data in peer-reviewed journal articles makes the data synthesis somewhat challenging. Each article has its own focus, and only limited information is available from each study. The types of emissions reported often vary, with studies variously presenting (and measuring) ebullitive versus diffusive emissions. In brief, the availability of emissions data depends largely on sampling methods of which the study proponents were familiar.

The collated data demonstrate that methane emission data from tropical climate are very limited, and most studies have been conducted in 'temperate' zones (Tables 2, 3 and 4). Methane emissions in diffusive form are the most available data compared to ebullitive emissions or combined emissions. In addition, the data highlight that more studies have been conducted in smaller waterbodies (<10 ha) compared to larger waterbodies (Tables 2, 3 and 4). This in part may be due to the understanding that small water bodies are important hotspots of methane production (Holgerson and Raymond, 2016) or it may be related to logistical challenges in measuring large waterbodies.

Water chemistry data are limited in the collected, emissions related articles. One third of these articles had no water chemistry data, while the remainder only reported certain parameters in terms of water chemistry. Physical parameters, such as water temperature, waterbody size or sampling depth, are also limited in the literature. In addition, the types of temperature measured in different articles are also different, for example, it could be measured in air, water, or sediment.

4.3 Methane emission levels

The collected methane emission data are summarised in Tables 2, 3 and 4 based on the data collected in Apx Tables C.1, C.2& C.3 to better understand the levels of methane emission from different climates and identify patterns between emissions and different sized areas. The emission

data in the appendices without the associated area information could not be included in these summarised tables.

Table 2 Minimum, maximum and median values of ebullitive methane emissions from different sized waterbodies within different climates

Area (ha)	Climate	Number of data	Minimum emissions ($mg/m^2/d$)	Maximum emissions ($mg/m^2/d$)	Median emissions ($mg/m^2/d$)
<0.1	Temperate	6	3.6	53.4	19
	Subtropical	5	143	1405	274
0.1-1	Temperate	8	2.5	484	83
	Subtropical	10	37	905	71
1-10	Temperate	9	7.3	300	27
	Subtropical	8	15	590	57
10-100	Temperate	7	0.3	337	3.7
100-1000	Temperate	4	62	444	84
1000-10000	Temperate	1	57	57	57
	Subtropical	1	23.3	23.3	23.3
	Tropical	2	12	65	39
>10000	Temperate	4	1.5	33.7	8
	Tropical	47	13	3635	548

Source: Burke et al. (2019); Grinham et al. (2018); Natchimuthu et al. (2014); Bastviken et al. (2004); van Bergen et al. (2019); Ortega et al. (2019); Casper et al. (2000); Mattson and Likens (1993); Strayer and Tiedje (1978); Schrier-Uijl et al (2010); Chau et al. (1977); Duchemin (2000); Keller and Stallard (1994); Xing et al. (2005)

Table 3. 3 Minimum, maximum and median values of diffusive methane emissions from different sized waterbodies within different climates

Area (ha)	Climate	Number of data	Minimum emissions ($mg/m^2/d$)	Maximum emissions ($mg/m^2/d$)	Median emissions ($mg/m^2/d$)
<0.1	Temperate	33	0.4	174	12
0.1-1	Arctic	1	1.28	1.28	1.28
	Temperate	26	0.37	219	18
	Subtropical	5	0.39	129	15
1-10	Arctic	2	3.7	8.5	6.1
	Temperate	47	0.23	1468	6.4
	Subtropical	2	22	44	33
	Tropical	5	10.6	595	14
10-100	Arctic	2	3.7	4.7	4.2
	Temperate	33	0.16	674	2.9
	Subtropical	1	1.41	1.41	1.41
	Tropical	4	7.7	11.6	11.2

100-1000	Arctic	2	5	16.4	10.7
	Temperate	12	0.2	82	1.9
	Subtropical	6	0.4	3.6	1.9
1000-10000	Temperate	11	0.07	21.5	4.4
	Subtropical	6	1.5	57.4	13.7
	Tropical	4	8	20	13.2
>10000	Temperate	13	0.47	53	4

Source: Bastviken et al.(2004); van Bergen et al. (2019); Casper et al. (2000); Goeckner et al (2022); Natchimuthu et al. (2014); Ortega et al. (2019); Peakcock et al (2019); Peakcock et al (2021); Thottathil and Prairie (2021); Riera et al. (1999); Smith and Lewis (1992); Webb et al (2019); West et al. (2016); Xiao et al. (2014); Kling et al. (1992); Audet et al. (2020); Bastviken et al. (2002); Huttunen et al. (2003); Michmerhuizen et al. (1996); Rudd and Hamilton (1978); Shaher et al. (2020); St Louis et al. (2000); Bastviken et al. (2010); Gorsky et al (2019); Santoso et al. (2021); Strayer and Tiedje (1978); Xiao et al. (2017) Schubert et al. (2010); Utsuumi et al. (1998b)

Table 4 Minimum, maximum and median values of combined methane emission from different sizes of waterbodies within different climates (combined emissions means the methane measurement includes both the ebullitive and diffusive emissions.)

Area (ha)	Climate	Number of data	Minimum emissions ($mg/m^2/d$)	Maximum emissions ($mg/m^2/d$)	Median emissions ($mg/m^2/d$)
<0.1	Temperate	3	23	227	133
0.1-1	Temperate	8	2.4	274	123
1-10	Temperate	22	0.3	503	7.5
	Tropical	3	23	353	263
10-100	Temperate	8	0.4	159	2.5
	Subtropical	2	20	2400	1210
	Tropical	4	34	200	109
100-1000	Subtropical	3	500	1200	800
1000-10000	Temperate	2	6	187	96.5
	Subtropical	2	6.7	71.3	39
	Tropical	2	90	92	91
>10000	Temperate	4	8	40	22
	Subtropical	4	3	13	6.5

Source: He et al. (2018); Huttunen et al. (2003); Morin et al. (2017); Natchimuthu et al. (2014); Ortega et al. (2019); Rabaey and Cotner (2022); Schmiedeskamp et al. (2021); West et al. (2016); Yuan et al. (2021) Bastviken et al. (2010); Sieczko et al. (2020); Schenk et al. (2021); Weyhenmeyer (1999); Bastviken et al. (2010); Martinez-Cruz et al. (2017); Hellsten et al., (1996) Bevelhimer et al. (2016); Zhao et al. (2013)

For each area category, the collected raw ebullitive, diffusive, and combined emissions data in Apx Tables C.1, C.2& C.3 from waterbodies in different climates are described as below and illustrated using a boxplot. The general knowledge of a boxplot is explained in Figure 9. The statistical tests, T test and ANOVA (Analysis of Variance) test, are used to compare two groups of data and three or more groups of data, respectively. P value resulted from T test and p-value resulted from ANOVA test less than 0.05 means significant different between groups.

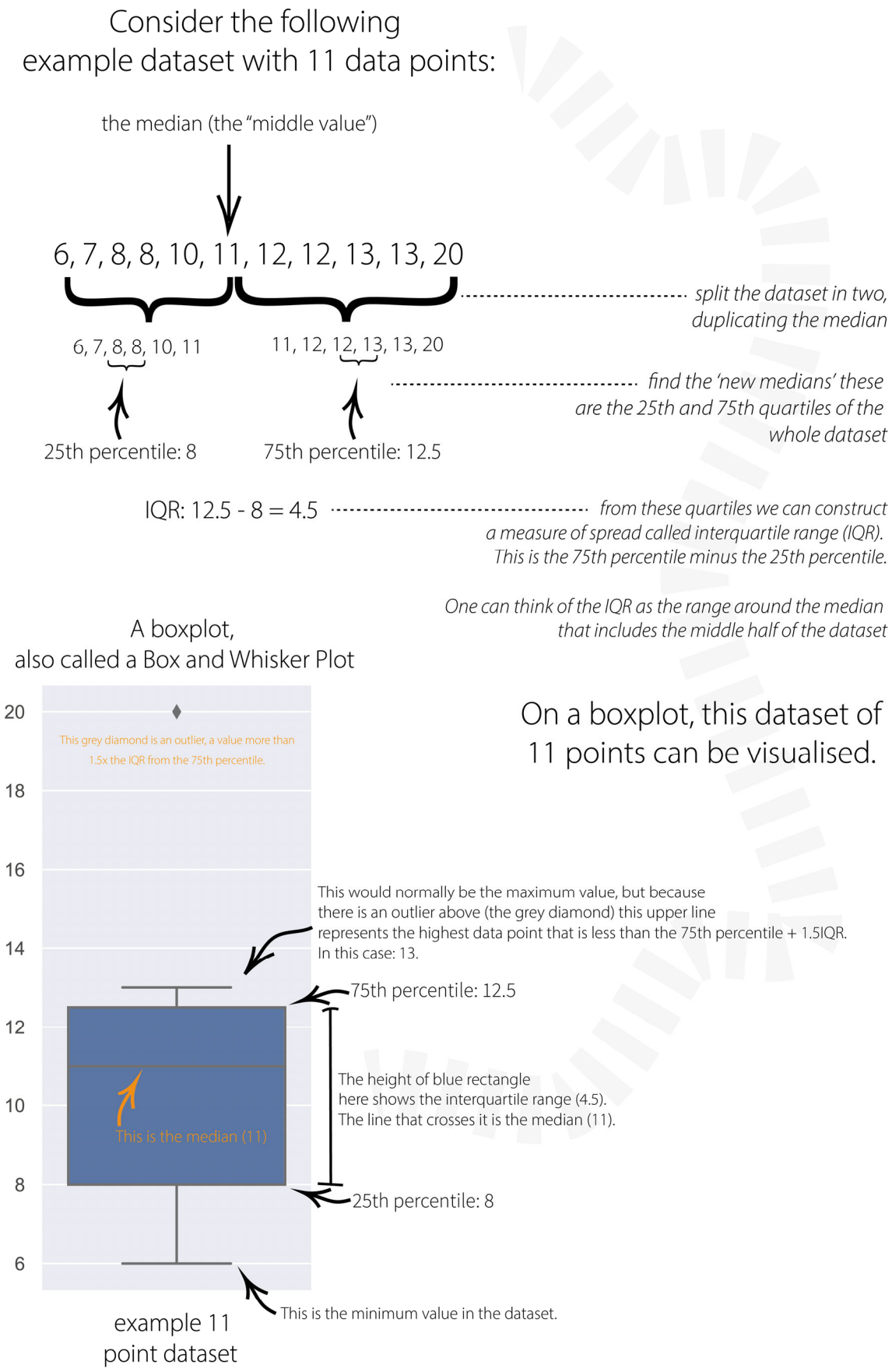


Figure 9 An illustration of general knowledge about a boxplot.

For small (<0.1), temperate waterbodies, the ebullitive, diffusive and combined methane emissions were 3.6 to 53.4 mg/m^2/d (n=5), 0.4 to 174 mg/m^2/d (n=33) and from 23 to 227 mg/m^2/d (n=3), respectively (Tables 2, 3 and 4). In contrast, the ebullitive methane emissions from same sized subtropical waterbodies were 143 to 1405 mg/m^2/d (n=6) (Table 3). The ebullitive methane emissions from subtropical waterbodies are significantly higher than all emissions from the temperate climate (Figure 10) with a p-value of 5.77e-5 from ANOVA test.

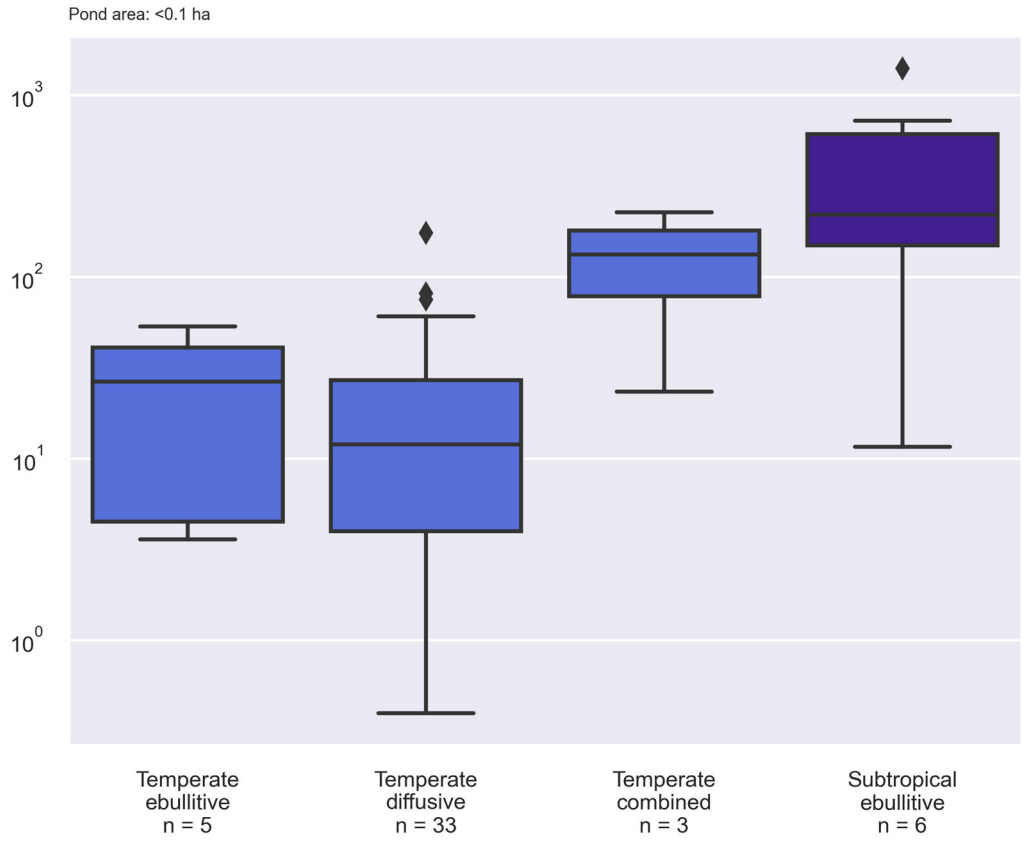


Figure 10 A boxplot showing different forms of methane emissions from waterbodies in temperate and subtropical climates with their areas less than 0.1 ha. The Y axis of the plot is in logarithmic scale.

For the moderately sized (0.1-1 ha), arctic waterbody, the diffusive methane emissions is 1.28 mg/m^2/d (n=1). For the same sized, temperate waterbodies, the ebullitive, diffusive and combined methane emissions were 2.5 to 484 mg/m^2/d (n=8), 0.37 to 219 mg/m^2/d (n=26) and 2.4 to 274 mg/m^2/d (n=11), respectively (Tables 2, 3 and 4). For the subtropical waterbodies of the same size, ebullitive and diffusive methane emissions were 37 to 905 mg/m^2/d (n=10) and 0.39 to 129 mg/m^2/d (n=5), respectively (Tables 2 and 3). The ebullitive and diffusive methane emissions are comparable in either subtropical or temperate climate (Figure 11).

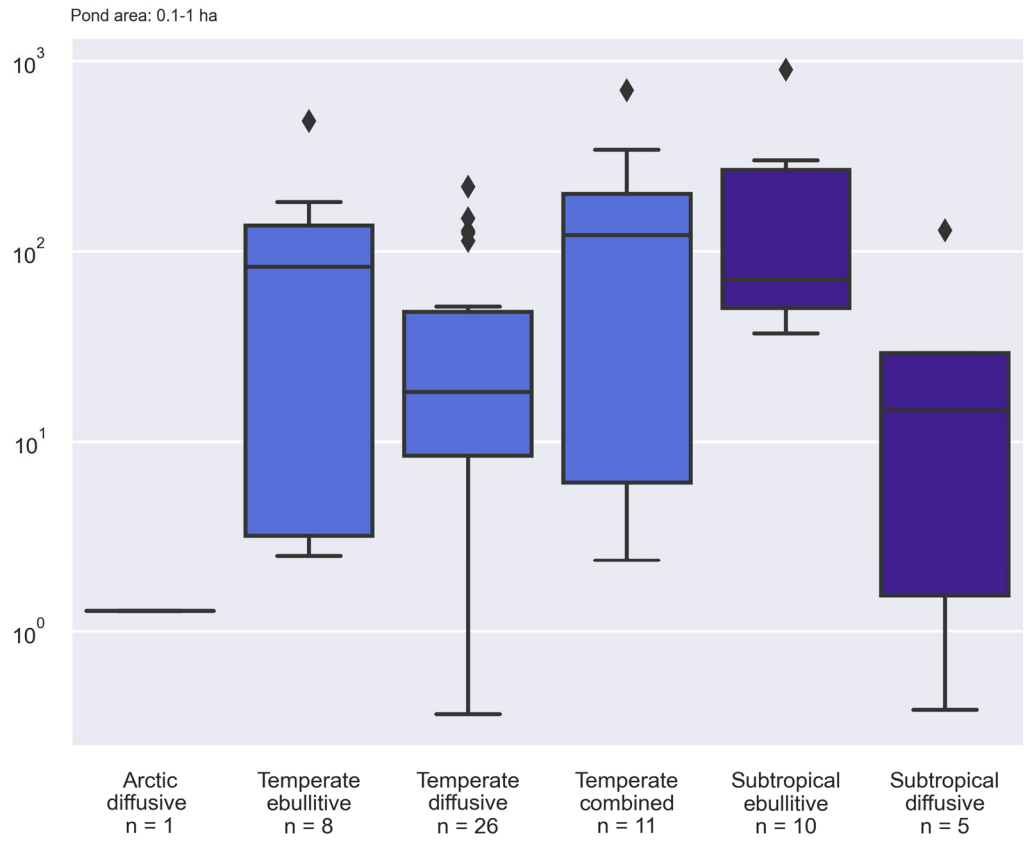


Figure 11 A boxplot showing different forms of methane emissions from waterbodies in arctic, temperate and subtropical climates with their areas between 0.1 and 1 ha. The Y axis of the plot is in logarithmic scale.

For moderately sized (1-10 ha), temperate waterbodies the ebullitive, diffusive and combined methane emissions were 7.3 to 300 mg/m^2/d (n=9), 0.23 to 1468 mg/m^2/d (n=47) and 0.3 to 503 mg/m^2/d (n=22), respectively (Tables 2, 3 and 4). For the same sized subtropical waterbodies, the ebullitive and diffusive methane emissions were 15 to 590 mg/m^2/d (n=8) and 22 to 44 mg/m^2/d (n=2), respectively (Tables 2 and 3). In contrast, the combined emissions from temperate and tropical waterbodies were 0.3 to 503 mg/m^2/d (n=22) and 23 to 353 mg/m^2/d (n=3), respectively (Table 4). The combined emission from tropical regions is not much higher compared to those from temperate regions (Figure 12, P value of 0.155 from T test) even if the median values are significantly different (Table 4). The ebullitive and diffusive methane emissions are not significantly different in different climates (Figure 12, P value of 0.38 from T test for ebullitive emissions from subtropical climate and *p*-value of 0.586 from ANOVA test for diffusive emissions from temperate climate).

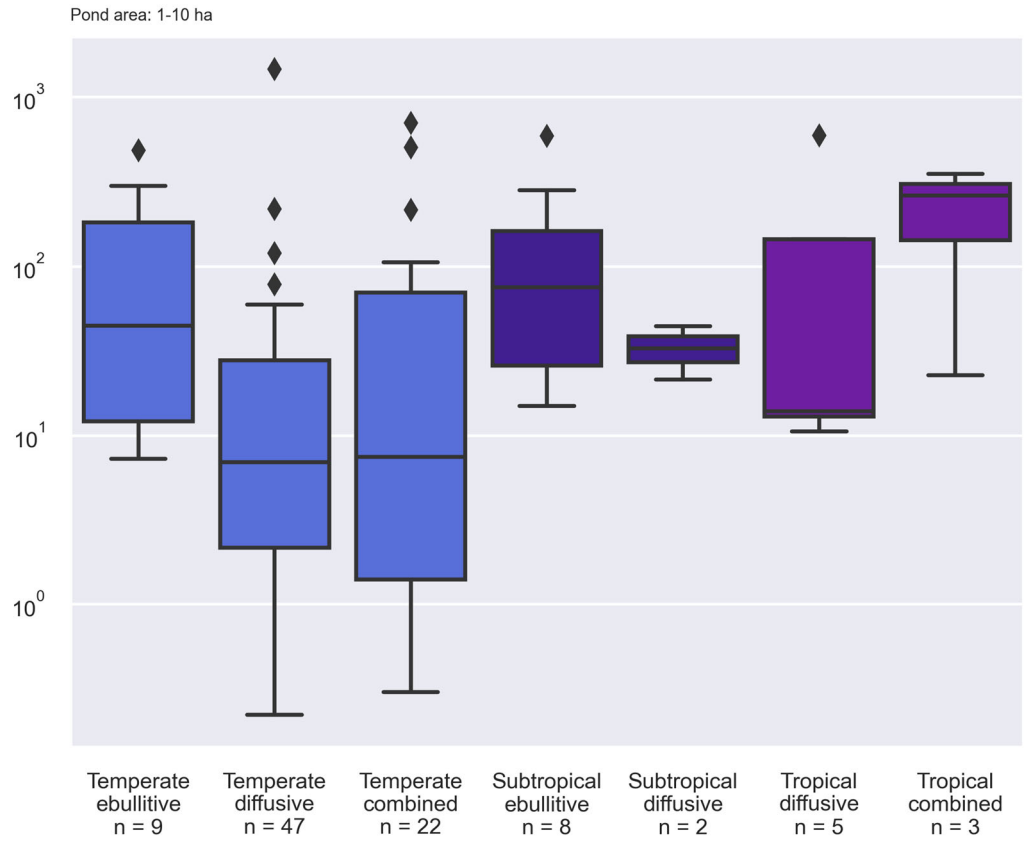


Figure 12 A box plot showing different forms of methane emissions from waterbodies in temperate, subtropical and tropical climates with their areas between 1 and 10 ha. The Y axis of the plot is in logarithmic scale.

For larger (10-100 ha), temperate waterbodies, the ebullitive, diffusive and combined methane emissions were 0.3 to 337 mg/m^2/d (n=7), 0.16 to 674 mg/m^2/d (n=33), and 0.4 to 159 mg/m^2/d (n=8), respectively (Tables 2, 3 and 4). Limited data are available from the subtropical regions in this category with the only diffusive methane emission being 1.41 mg/m^2/d (Table 3) and two combined emissions data being 20 and 2400 mg/m^2/d (Table 4). The diffusive and combined methane emissions from tropical waterbodies were 7.7 to 11.6 mg/m^2/d (n=4) and from 34 to 200 mg/m^2/d (n=4), respectively (Tables 3 and 4). The much higher combined methane emission in subtropical climates compare to those from the tropical and temperate climates (Figure 13, p-value: 0.041 from ANOVA test) was from rivers in Mexico City and appears to represent something of an outlier that may reflect local conditions. For details of this study please see: Martinez-Cruz et al. (2017).

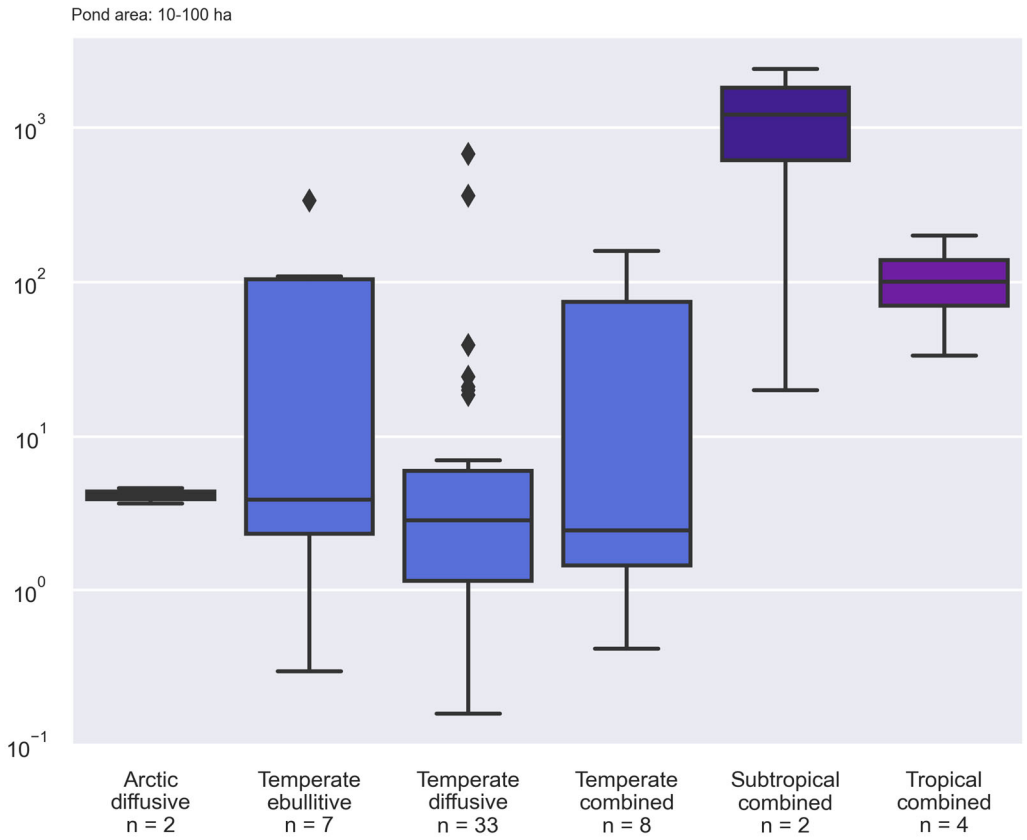


Figure 13 A box plot showing different forms of methane emissions from waterbodies in arctic, temperate, subtropical and tropical climates with their areas between 10 and 100 ha. The Y axis of the plot is in logarithmic scale.

For larger (100-1000 ha), arctic waterbodies, the diffusive methane emissions were 4.97 to 16.36 mg/m^2/d (n=2) (Table 3). For the same sized, temperate waterbodies, the ebullitive and diffusive methane emissions were 62 to 444 mg/m^2/d (n=4) and 0.2 to 82 mg/m^2/d (n=12), respectively (Tables 2 and 3). For subtropical waterbodies of the same size, the diffusive and combined methane emissions were 0.4 to 3.6 mg/m^2/d (n=6) and 500 to 1200 mg/m^2/d (n=3) (Tables 3 and 4). The high combined methane emissions from the subtropical climate region (Figure 14, *p*-value of 2.7e-8 from ANOVA test) were emitted from canals, chinampas and lakes in Mexico City (Martinez-Cruz et al., 2017).

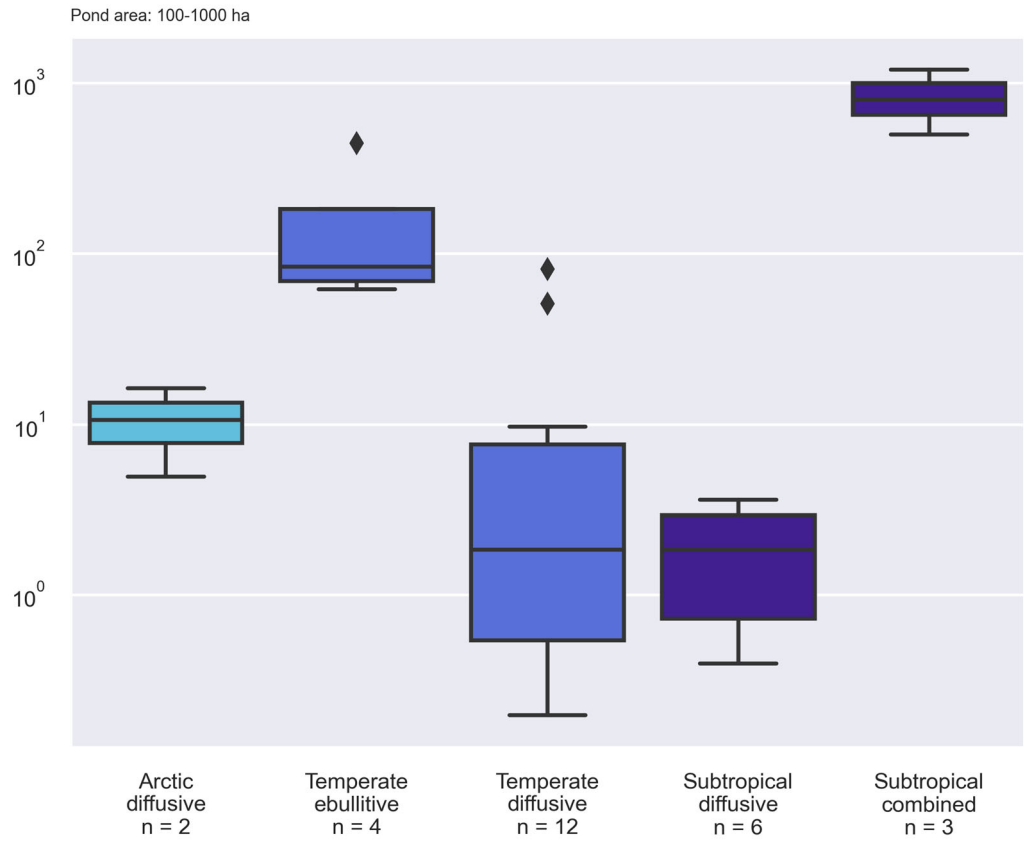


Figure 14 A box plot showing different forms of methane emissions from waterbodies in arctic, temperate and subtropical climates with their areas between 100 and 1000 ha. Y axis of the plot is in logarithmic scale.

For larger (1000-10000 ha), temperate waterbodies, the ebullitive, diffusive and combined methane emissions were 57 mg/ m^2/d (n=1), 0.07 to 215 mg/m^2/d (n=11) and 6 to 187 mg/m^2/d (n=2), respectively (Tables 2, 3 and 4). For the same sized ponds in the subtropics, the ebullitive and diffusive methane emissions were 23.3 mg/m^2/d (n=1) and in a range from 0.4 to 3.6 mg/m^2/d (n=6), respectively (Tables 3 and 4), while in tropical waterbodies, the ebullitive, diffusive and combined methane emissions were 12 to 65 mg/m^2/d (n=2), from 7.8 to 20 mg/m^2/d (n=4) and from 90 to 92 mg/m^2/d (n=2), respectively (Tables 2, 3 and 4). In both temperate and tropical climates, it is obvious that the ebullitive and combined emissions are much higher than the diffusive emissions (Figure 15, *p*-value: 0.029 (temperate) and 0.0098 (tropical) from ANOVA test).

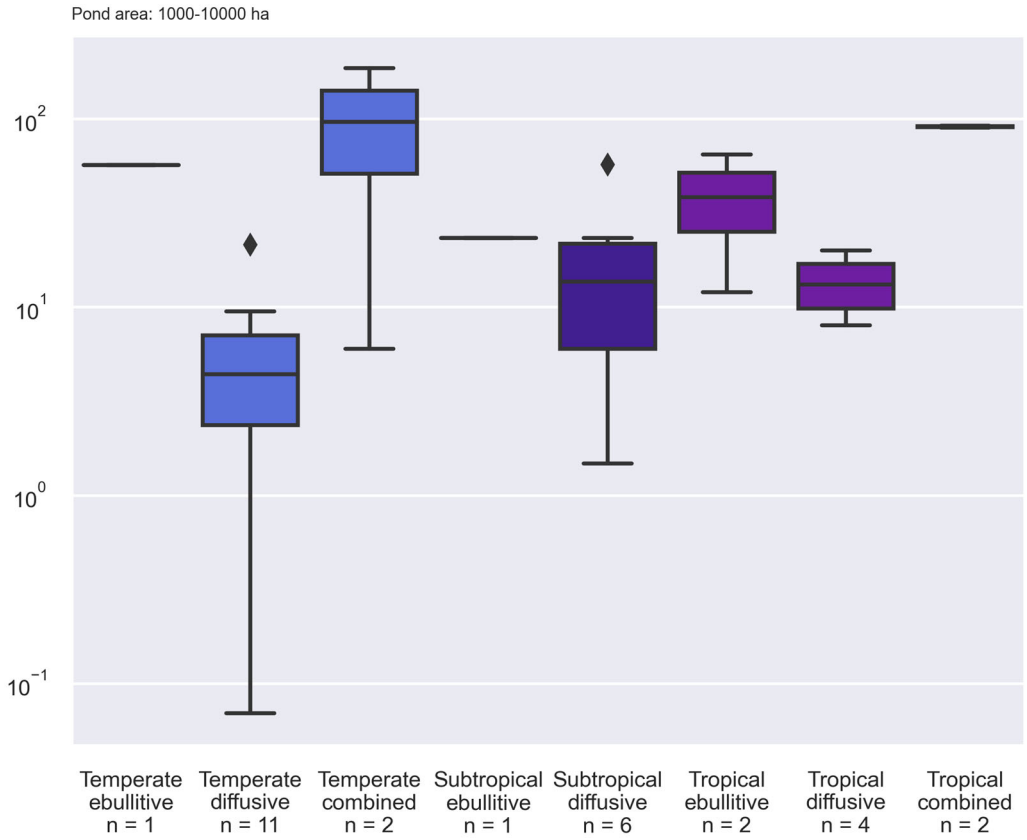


Figure 15 A boxplot showing different forms of methane emissions from waterbodies in temperate, subtropical and tropical climates with their areas between 1000 and 10000 ha. The Y axis of the plot is in logarithmic scale.

For the largest (>10000 ha), temperate waterbodies, the ebullitive, diffusive and combined methane emissions were 1.5 to 33.7 mg/m^2/d (n=4), 0.47 to 53 mg/m^2/d (n=13) and 8 to 40 mg/m^2/d (n=4), respectively (Tables 2, 3 and 4). For the same sized ponds in the subtropics, the combined methane emissions were 3 to 13 mg/m^2/d (n=4) (Table 4) while in tropical regions, the ebullitive methane emissions were 13 to 3635 mg/m^2/d (n=47) (Table 2). The ebullitive methane emissions (Figure 16) from the tropical region are significantly higher than all forms of emissions from the temperate regions and combined emissions from the tropical regions (Figure 16, *p*-value: 4.23e-4 from ANOVA test).

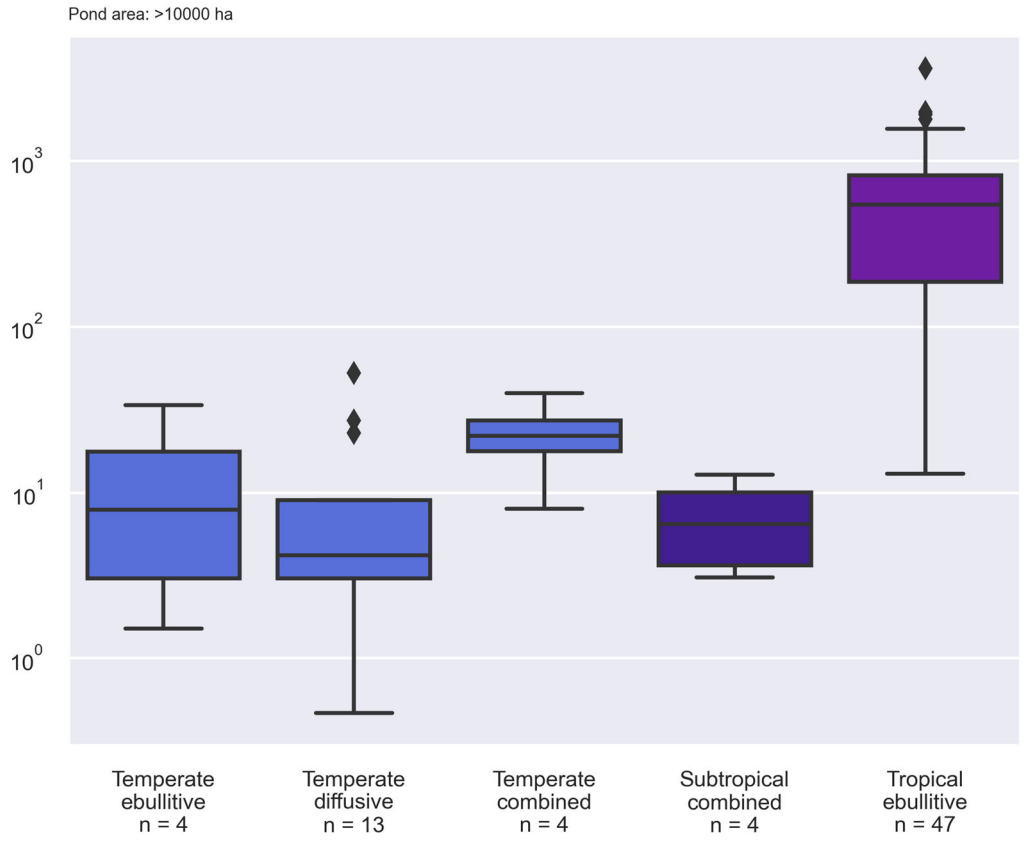


Figure 16 A boxplot showing different forms of methane emissions from waterbodies in temperate and subtropical, tropical climates with their areas larger than 10000 ha. The Y axis of the plot is in logarithmic scale.

4.4 Ebullitive vs. diffusive emission

Literature review showed that there are only a handful of published studies that have measured diffusive and ebullitive emissions simultaneously, and these studies report a very wide range of ebullitive : diffusive emissions ratios ranging from 0.2 to 31 (Bastviken et al., 2004; Casper et al., 2000; Duchemin et al., 2000; Herrero Ortega et al., 2019; Huttunen et al., 2003; Natchimuthu et al., 2014; Rabaey and Cotner, 2022; Strayer and Tiedje, 1978; van Bergen et al., 2019). This is not surprising since methane ebullition is known to be episodic and highly spatially heterogeneous (Delsontro et al., 2015, 2011; Grinham et al., 2011). The overall ebullitive methane emissions from the collected data, however, is significantly higher than the diffusive methane emissions in tropical, subtropical and temperate regions (Figure 17) with P values from T test being 6.74e-8, 0.00139 and 0.0417, respectively.

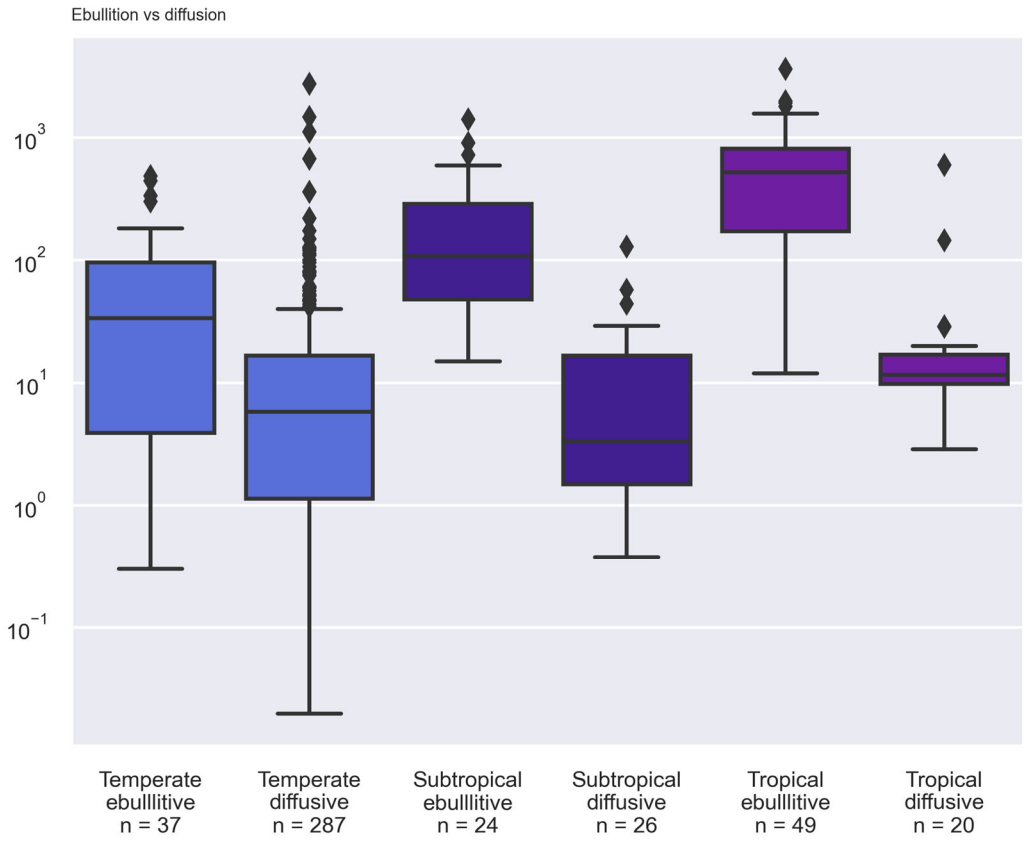


Figure 17 A boxplot showing ebullitive and diffusive methane emissions from waterbodies in temperate, subtropical and tropical climates. The Y axis of the plot is in logarithmic scale.

4.5 Driving factors of methane emissions

4.5.1 Temperature/climate

Methane in industry ponds can have two possible sources. The first is biologically generated methane which is produced by microbial degradation of organic matter (see: Section 2.2). The second is the amount of geological methane that is carried over post-separator (see: Introduction). Prior to a discussion about how the various pathways of methane emissions are correlated (or not) with temperature it is worth considering the following:

1) increasing temperature results in an increase in microbial activity (Peacock et al., 2021; St. Louis et al., 2000). For example, the work of Walter and Heimann (2000) suggests that every 1° increase in soil temperature results in a 20% increase in methane emissions from that soil. This is counterbalanced by an increase in growth of methanotrophs (at least in natural environments) which consume the increasing amount of methane (see Section 2.4).

2) increasing temperature decreases methane solubility in water. In practice this phenomenon means that at warmer temperatures, more of the methane is likely to form bubbles and therefore contribute to ebullitive emissions (Walter et al., 2007).

3) For biologically generated methane, the three pathways: ebullitive, diffusive and plant-mediated, all have the same source. The methane is being produced by methanogens in the anoxic zones within water bodies. Diffusive methane leaves this environment after dissolving into the

water, ebullitive methane leaves as bubbles and plant-mediated methane travels through plant tissues to escape the water column.

The collected ebullitive and combined methane emissions in this study are positively correlated with increasing temperature (r = 0.38 and 0.55 respectively; Figures 18a & 18c). The same trend is evidenced by the ebullitive data and climate (Figure 19). This has been previously demonstrated by others, for example, Aben et al. (2017) revealed strong positive relationships between ebullitive methane emissions and temperature for a wide range of shallow, freshwater ecosystems in Asia, North America and Europe. Similarly, Baron et al. (2022) reported that methane fluxes increased exponentially with temperature. In addition, the seasonal trend of the ebullitive methane emissions with highest ebullitive methane emissions in the summer also indicated the relationship between emissions and temperature (Grinham et al., 2018; Natchimuthu et al., 2014; Peacock et al., 2021; van Bergen et al., 2019).

The magnitude of temperature dependence of methane ebullition was also observed different in different freshwater ecosystems (Aben et al., 2017) which is related to differences in quantity and quality of sediment organic matter (Duc et al., 2010; Maeck et al., 2013; Schwarz et al., 2008; Sobek et al., 2012; W. E. West et al., 2012; West et al., 2015), sediment structure (Liu et al., 2016), different availability of nutrients, oxygen and alternative electron acceptors (Duc et al., 2010) and the types of temperature measured which could be air temperature (Baker-Blocker et al., 1977; Gao et al., 2013), sediment temperature (DelSontro et al., 2016; Walter and Heimann, 2000; Wik, 2016; Wik et al., 2014) or water temperature (Aben et al., 2017; Wilkinson et al., 2015).

Experiments by Aben et al. (2017) demonstrated that 4 °C increases resulted in a 51% increase in methane ebullition, but no effect was noted on diffusive emissions. This is likely due to the longer residency times of dissolved methane in the water column, which results in more opportunities for this methane to be lost to methanotrophy. Further, this likely contributes to observations in this study that diffusive methane emission does not correlate with either temperature or climatic zone (Figures 18b and 19). This has been observed previously in other research, for example, Hao et al. (2021) found that no significant positive correlation between the diffusive methane emissions and water temperature, on the contrary, diffusive methane emissions decreased while water temperature increase.

That temperature drives methane emissions through ebullition creates a kind of positive feedback loop, where increasing global temperatures increase emissions, which in turn drive increasing global temperatures (Aben et al., 2017).

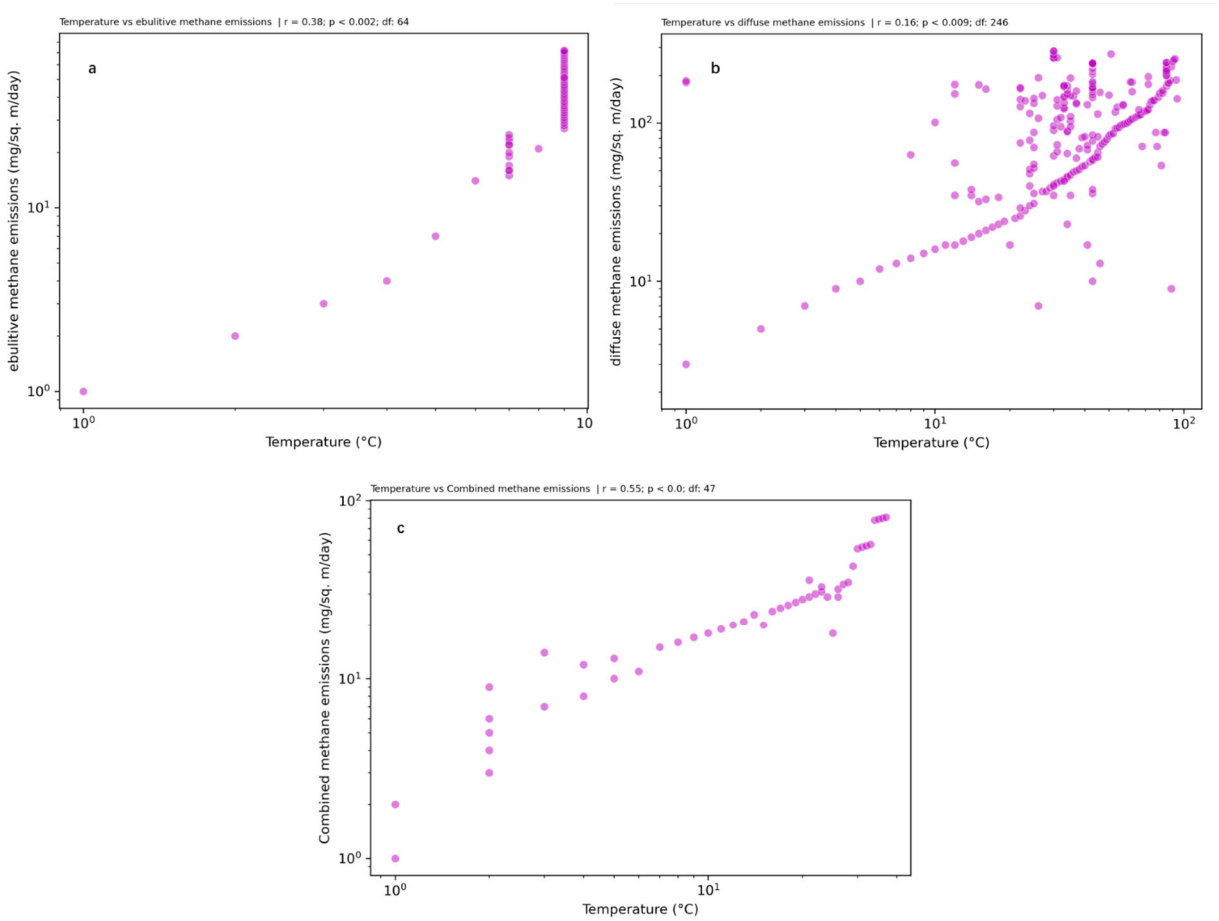


Figure 18 Scatter plots of ebullitive (a), diffusive (b) and combined (c) methane emissions versus temperature with logarithmic scale. Higher r value shows more correlated between emissions and temperature. Both X and Y axes of the plots are in logarithmic scales.

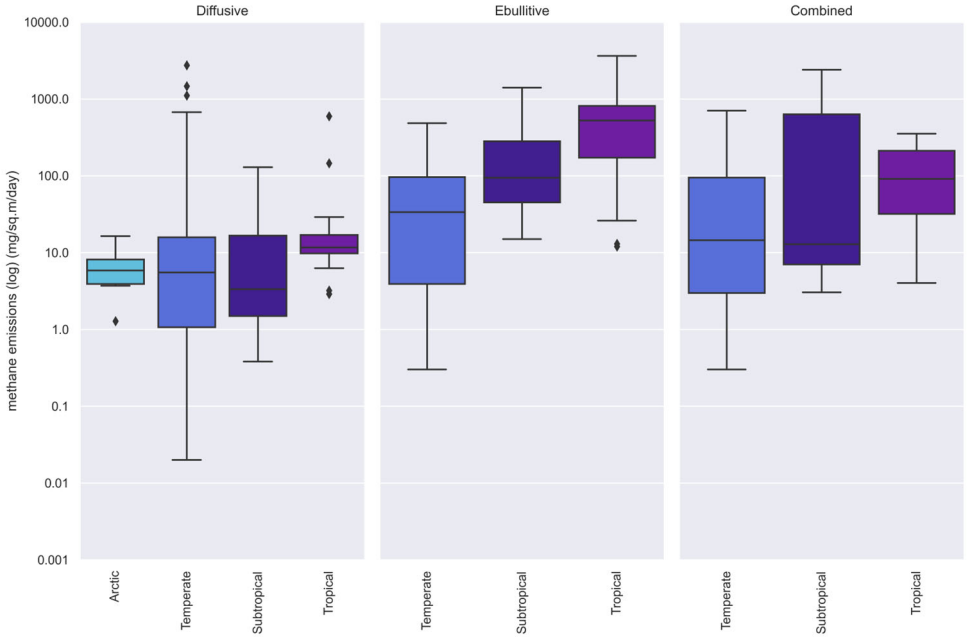


Figure 19 Boxplots showing comparison of diffusive (left), ebullitive (middle) and combined (right) methane emissions in different climates

4.5.2 Nutrients

Nutrients drive increases in microbial activity by removing constraints on microbial growth. This can be illustrated well using an analogy. If one collects sticks and places them in a pile to compost, in a year, one will still have a pile of sticks. In contrast, adding manure or fresh green leaves to the pile with the sticks will allow the sticks to be composted in a relatively short period of time.

In this analogy, there is no shortage of carbon, hydrogen or oxygen in the pile; dry sticks are essentially just cellulose and lignin (polymers of carbon hydrogen and oxygen). Microbes, however, are unable to act on the sticks due to an absence of important "*non-stick*" nutrients.

In particular, microbes need: nitrogen – to make proteins and nucleic acids (DNA or RNA), phosphorus – to make nucleic acids, phospholipids for the cell membrane, and energy-carrying molecules, sulfur – for a few amino acids (components of proteins) and several vitamins; along with sodium, potassium and magnesium which are variously important for cell function. Collectively these compounds are terms macronutrients (as they are needed in relatively 'large' amounts). Microbes also require a host of micronutrients, but as these are needed at very low levels, they frequently do not limit microbial activity.

Phosphorus

Total phosphorous in the natural and anthropogenically constructed waterbodies examined in this study range from 0.25 to 6480 µg/L with a median value of 44 ug/L. Total phosphorous concentrations have positive correlations with all forms of methane emissions with r values being 0.45, 0.65 and 0.35 for ebullitive, diffusive and combined methane emissions, respectively (Figure 20). Total phosphorous has been consistently reported to have a positive relationship with ebullitive methane emissions from waterbodies such as ditches and anthropogenically built ponds (Audet et al., 2020; Beaulieu et al., 2019; DelSontro et al., 2016; M. Peacock et al., 2019; Peacock et al., 2021). Experimental data by Davidson et al. (2018), however, showed that nutrient enrichment had no effect on diffusive methane emissions, instead, ebullitive emissions were correlated with nutrient enrichment.

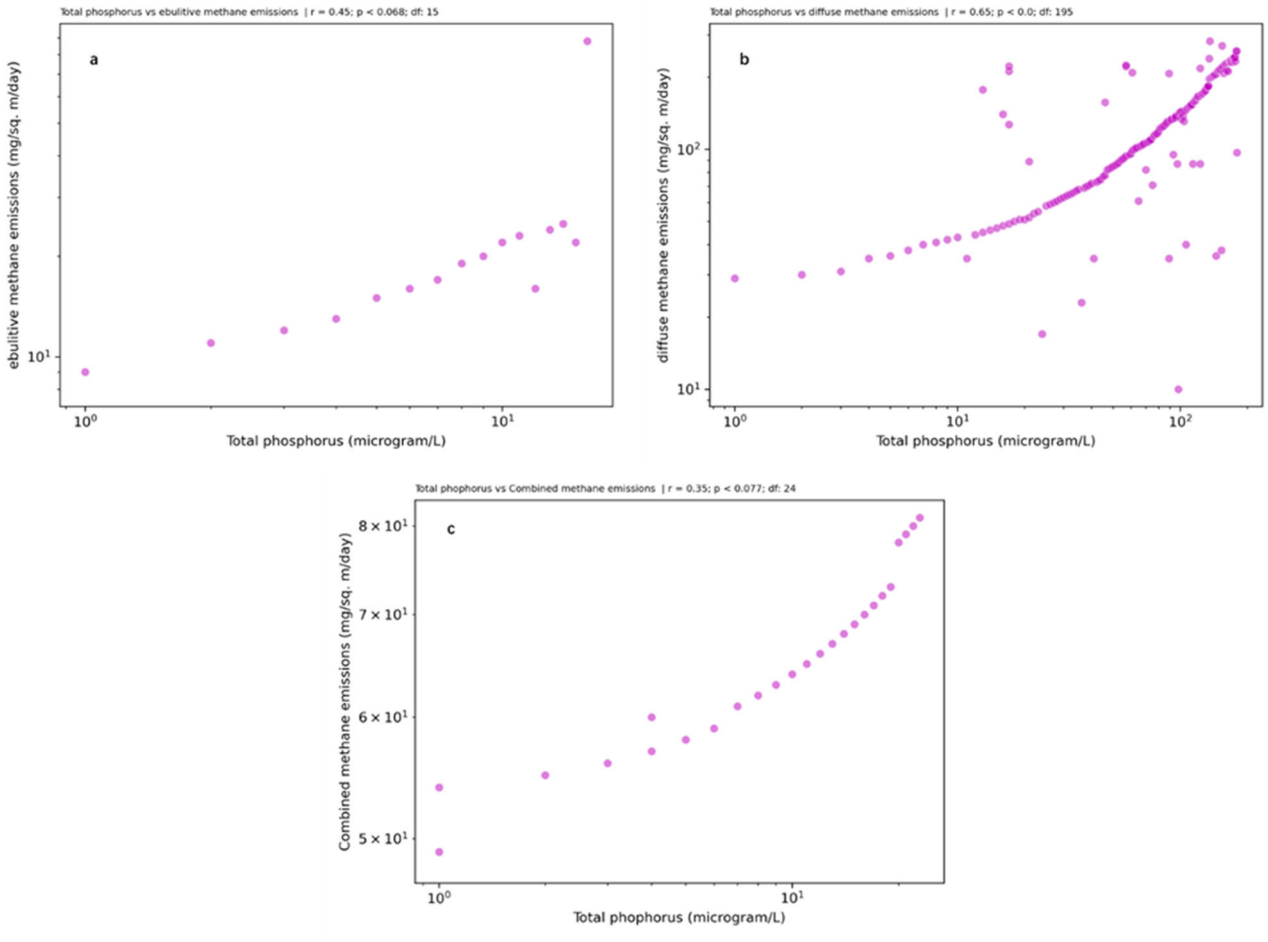


Figure 20 Scatter plots of total phosphorous versus ebullitive (a), diffusive (b) and combined (c) methane emissions with logarithmic scale showing all forms of methane emissions are positively correlated to total phosphorous. Both X and Y axes of the plots are in logarithmic scales except the Y axis of Figure 20c.

Nitrogen

In the current study, no correlation was observed between total nitrogen and methane emissions (Figure 21). In broad terms this is counter intuitive as nitrogen limited environments would necessarily produce less methane through constraints on microbial activity. This can in part be explained by plentiful atmospheric nitrogen, which can be fixed by microbes into usable forms rendering nitrogen infrequently limiting. Sometimes, however, more complex interactions are possible, and the chemical species of nitrogen may impact methanogenesis. For instance, in oxic zones in aquatic settings ammonia/ammonium (NH_3/NH_4^+) and nitrite (NO_2^-) are microbially oxidised to nitrate (NO_3^-; Figure 22) and it is nitrate that is often the dominant form of soluble nitrogen in aquatic systems and this form of nitrogen can interact with methanogenic processes. It does this through shifting the microbial community from one where methanogenesis is the dominant process, to one where nitrate reduction is the dominant process in the anoxic zones (Figure 22). In nitrate reducing microbial communities, nitrate is reduced via nitrite, then nitric oxide, nitrous oxide and then nitrogen gas. This shift is too complex to explain in detail. In brief, however, microbes gain more energy from reducing nitrate (to dinitrogen gas) than carbon dioxide (to methane) and this energetic difference disfavours methanogenic processes. Experimental evidence for such processes can be seen in Audet et al. (2020) who reported a negative correlation between NO_3^- and CH_4 concentrations.

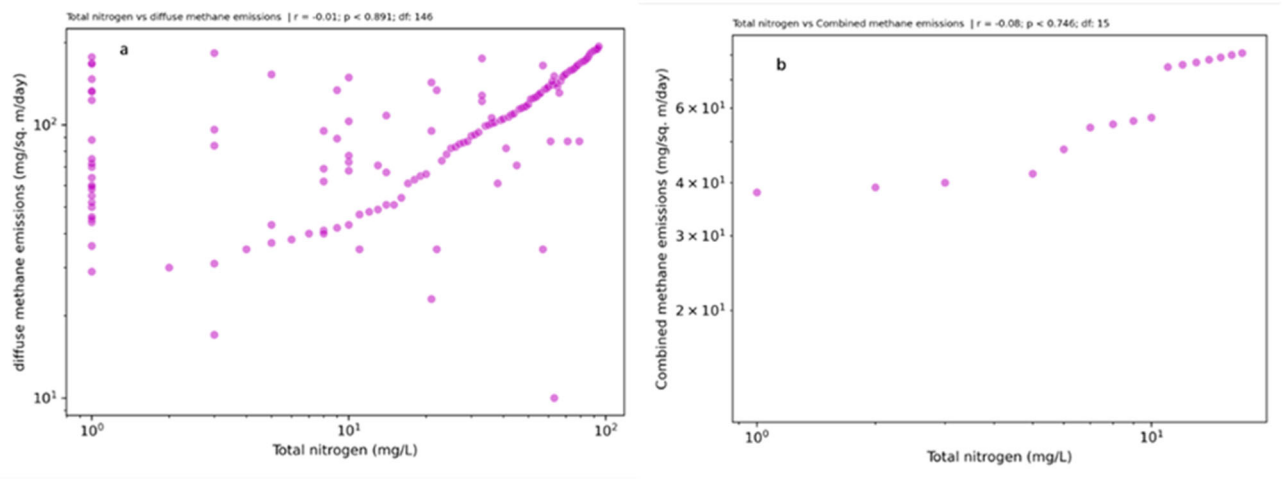


Figure 21 Scatter plots of total nitrogen versus diffusive (a) and combined (b) methane emissions with logarithmic scale showing that total nitrogen is not correlated to methane emissions as evidenced by the r values being very low. Both X and Y axes of the plots are in logarithmic scales except the Y axis of Figure 22b.

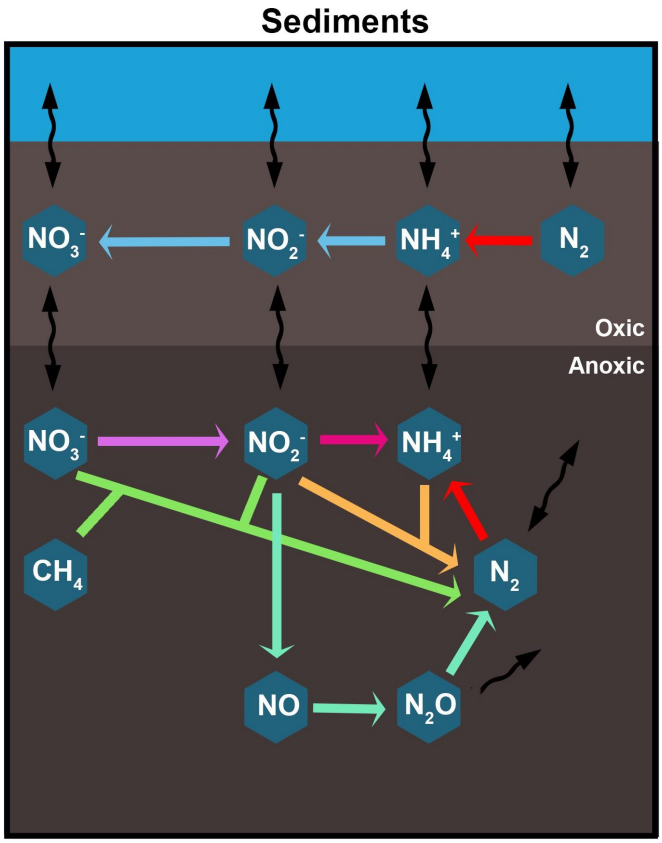


Figure 22 Nitrogen transformations in sediments in oxic and anoxic zones. N_2 = di-nitrogen gas, NH_4^+ = ammonium, NO_2^- = nitrite, NO_3^- = nitrate, N_2O = nitrous oxide, NO = nitric oxide. Figure represents a crop of an original image by Silvia Pajares and Ramiro Ramos reproduced here under a Creative Commons 4.0, Attribution-only license. The original can be found in Pajares and Ramos (2019).

Carbon

Dissolved organic carbon (DOC) in the non-CSG waterbodies collated in this study range from 0.74 to 146 mg/L. Here, DOC showed a slight positive correlation with combined methane emissions with r value of 0.32 but no correlation with ebullitive or diffusive methane emissions (Figure 23).

This is unexpected as DOC represents a key pool of mobilisable carbon in waterbodies and may be an artefact of the studies examined here or may be due to DOC being the most labile pool of carbon (i.e. this pool is readily consumed by microbes). Supporting such a contention, M. Peacock et al. (2019) reported significant positive correlations between dissolved CH_4 and total organic carbon. Deemer & Holgerson (2021) also suggested that DOC is an important predictor for methane emissions. Further, it seems only logical that anaerobic decomposition of organic matter derived from allochthonous sources provides addition DOC in the waterbodies which can then act as a substrate for variously microbial processes, which, eventually, result in increases in methanogensis in anoxic zones (Deemer and Holgerson, 2021; Grinham et al., 2018; Wik, 2016).

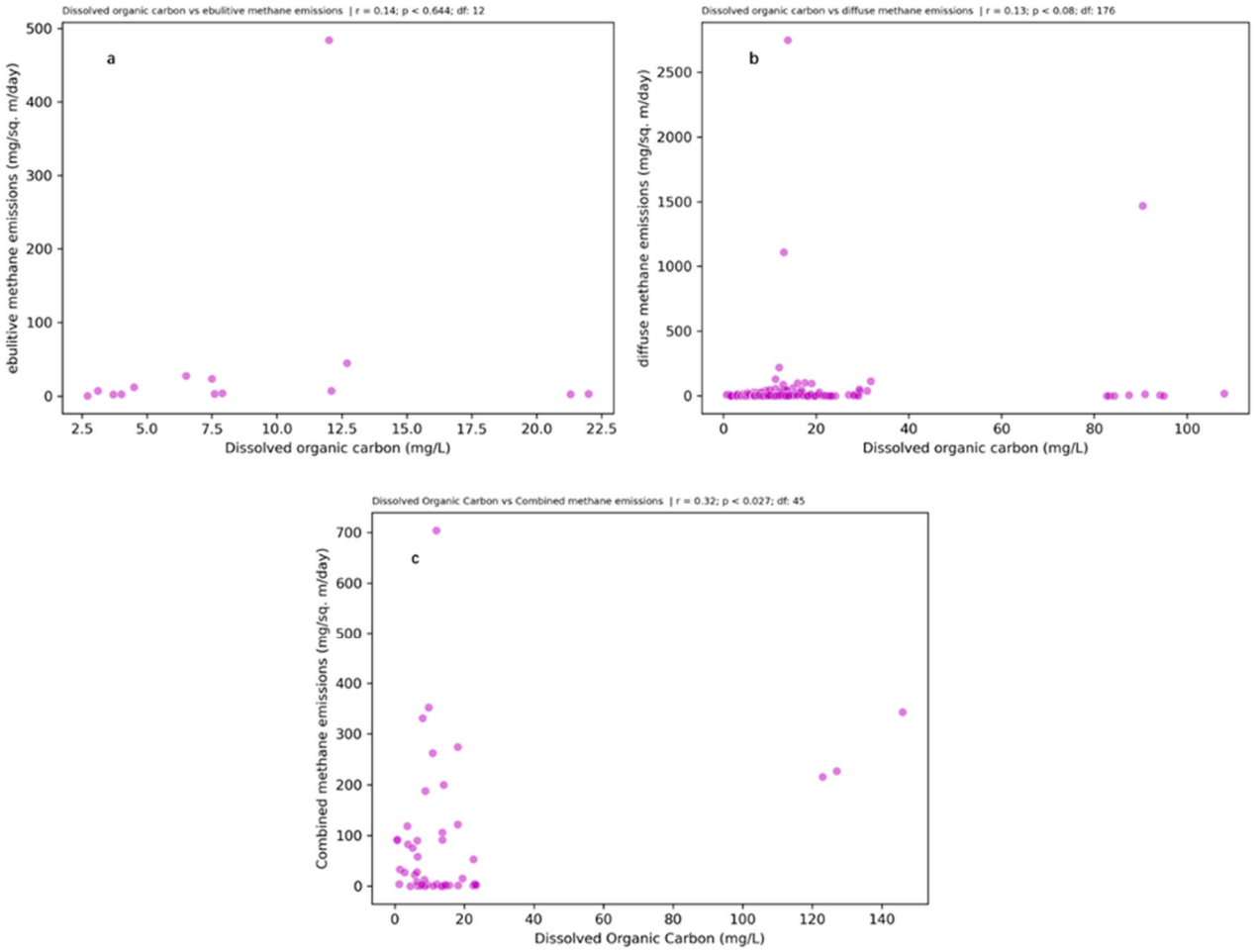


Figure 23 Scatter plots of dissolved organic carbon versus ebullitive (a), diffusive (b) and combined (c) methane emissions showing the combined methane emissions are slightly positively correlated to dissolved organic carbon but not the ebullitive and diffusive emissions.

More complex nutrient interactions

In broader terms, nutrient rich waters are an issue unto themselves. Waterbodies where nutrients are high are frequently subject to the formation of algal blooms, which in turn, can access large amounts of previously unavailable sources of carbon (e.g. bicarbonates). Subsequently these algae die (as they typically grow to consume the excess nutrients in a boom-bust, ruderal lifecycle) and release their nutrients in the water which promotes further blooms, all of which are providing carbon and other macronutrients that can promote methane generation by methanogens (Audet et al., 2020; W. E. West et al., 2012; West et al., 2015).

4.5.3 Waterbody area

Waterbody area has a relatively complex relationship with emissions. It is self-evident that a 1000 ha water body has more total emissions than a 0.1 ha waterbody (see for example Bastviken et al. (2004)). The rate of emissions, however, on a per m^2 basis is considerably higher for small water bodies, which have been discussed previously (see Section 4.2). This means that small water bodies are disproportionately contributing to emissions. For instance, ten 1 ha waterbodies produce markedly more methane than a single 10 ha water body, making small water bodies hot spots for greenhouse gas emissions (Audet et al., 2020; Grinham et al., 2018; Holgerson and Raymond, 2016; Peacock et al., 2021; van Bergen et al., 2019). This phenomenon is reflected in the data collated in the present study, which show small to medium size waterbodies (<10 ha) emit methane emissions at a higher rate than larger waterbodies, especially ebullitive emissions in the subtropics (Table 1, Figure 24a) and diffusive emissions in temperate regions (Figure 25c). A Queensland example is that small waterbodies ranging from 0.02 to 5.6 ha emitted ebullitive emissions from 17 to 1405 mg/m^2/d in average value and a maximum value at 5425 mg/m^2/d from a Weir (Grinham et al., 2018). Hence small waterbodies should not be neglected in terms of accounting for regional or global methane emissions.

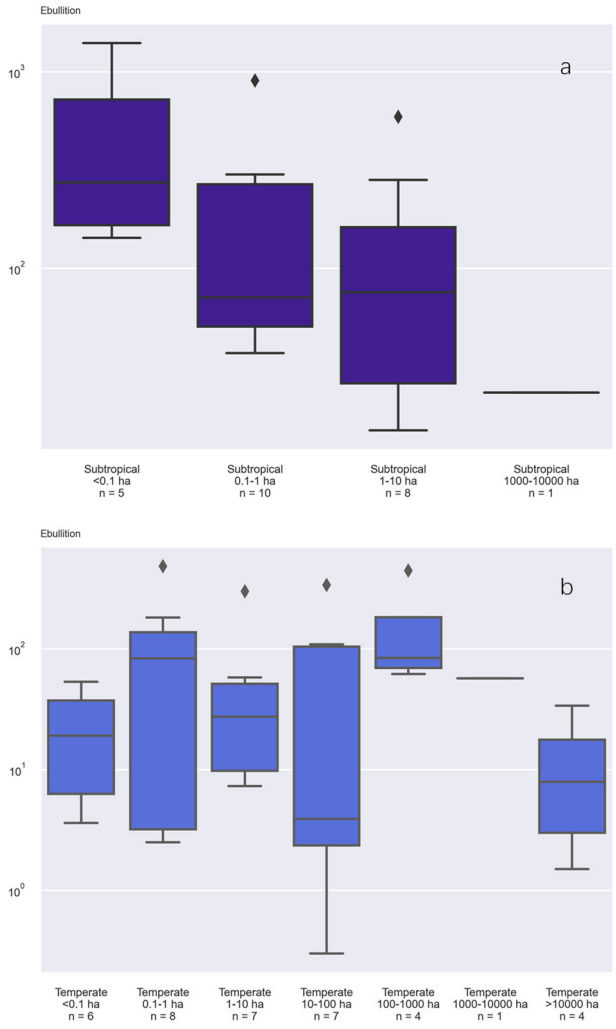


Figure 24 Boxplots showing ebullitive emissions from different sized waterbodies in subtropical regions (a) and temperate regions (b). The Y axes of the plots are in logarithmic scales.

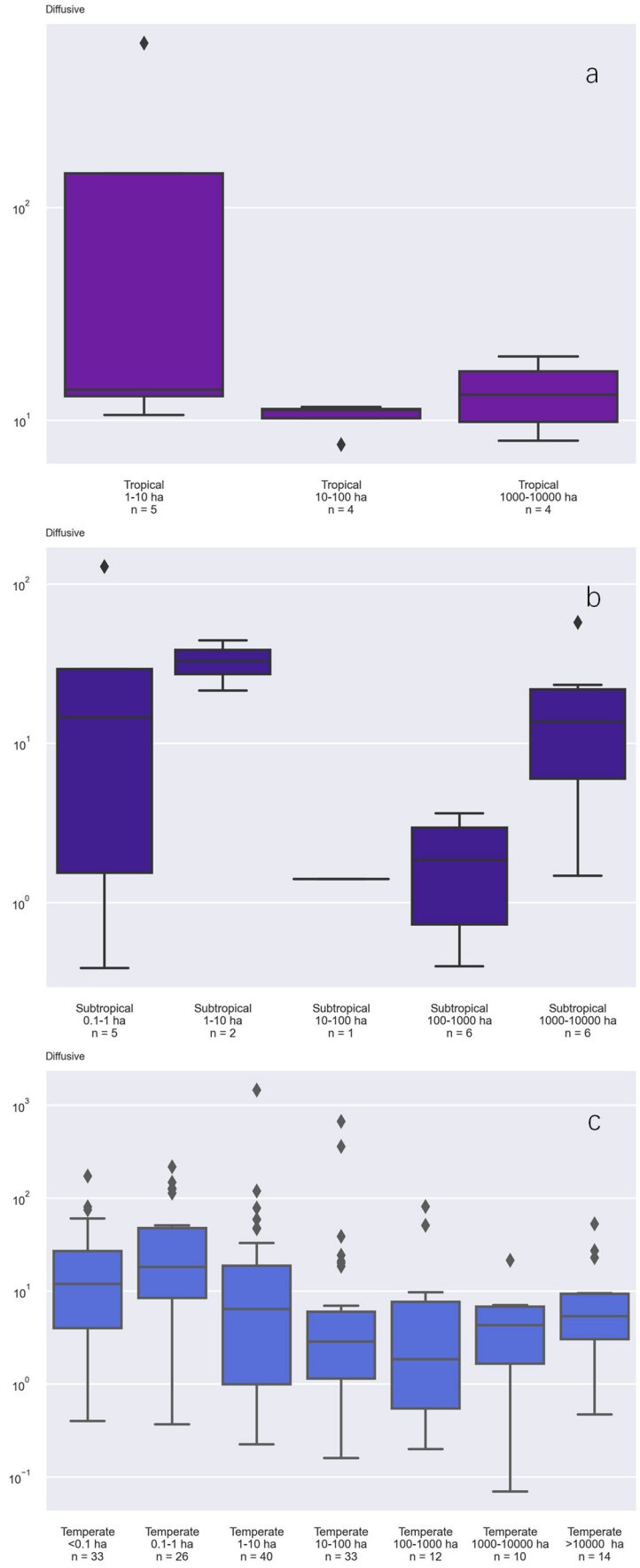


Figure 25 Boxplots showing diffusive emissions from different sized waterbodies in tropical (a), subtropical (b) and temperate (c) regions. The Y axes of the plots are in logarithmic scales.

4.5.4 Waterbody depth

Waterbody depth is a characteristic that interacts with a number of other attributes. In natural settings, pond size is often related to depth, with larger ponds often also being deeper. In addition, temperature effects are more pronounced in shallow depths as the sediment temperature can play a major role in microbial activity and in solubility of methane.

This behaviour can be observed by notably high ebullition rates often occurring in shallow waters (less than 4-6 m; Bastviken et al., 2004; DelSontro et al., 2016; Grinham et al., 2018; Joyce and Jewell, 2003; Keller and Stallard, 1994; Natchimuthu et al., 2014; West et al., 2016; Wik, 2016). This is likely due to limited stratification, sediment temperature being strongly related to atmospheric temperature (Aben et al., 2017), and direct solar warming of the sediments (Thornton et al., 2015; Wik et al., 2014). Also, shallow depths tend to have lower pressures compared to the deeper depths, further accelerate methanogenesis.

As previously discussed methanotrophy plays an important role in modulating the relationship between methane being produced and methane being emitted. The impact of this process is likely more pronounced in deeper lakes as bubbles have a longer residency time (Bastviken et al., 2004; Deemer et al., 2016; DelSontro et al., 2016; Herrero Ortega et al., 2019; Keller and Stallard, 1994; Peacock et al., 2021; Webb et al., 2019). Indeed, collated data in the present study show that depth is negatively related with emissions (r=-0.31,Figure 26a), a cautionary note being that most data in Figure 26a comes from a single study of the Gatun Lake, Panama (Keller and Stallard, 1994). In this study we were unable to observe a relationship between diffusive, combined emissions and depth (Figure 26b and 26c).

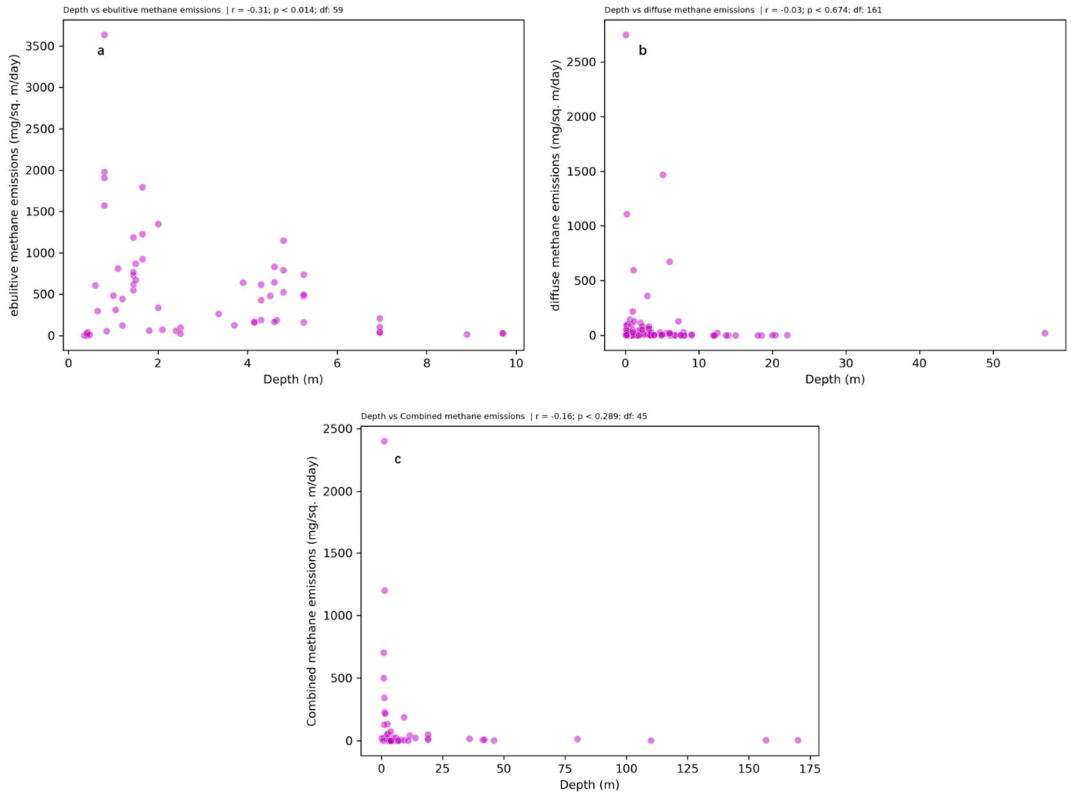


Figure 26 Scatter plots showing relationships between waterbody depth and ebullitive emissions (a), diffusive emissions (b) and combined emissions (c).

4.5.5 pH

Limited data collected in this study show that increasing pH (3.8-9) has a slight positive correlation with ebullitive methane emissions (Figure 27a) having a r value of 0.43. But neither diffusive methane emissions nor combined methane emissions correlate with increasing pH (Figure 27b and 27c). While the majority of the methanogens prefer neutral pH environments (pH 6-8) there are a host of methanogens that are specialised to acidic and alkaline environments. For example, shallow peatlands in Arctic region with low pH (3.8 to 5.9) were shown to emit considerable methane (20 mg/m^2/d ebullitive methane; Burke et al., 2019). Indeed, low pH environments like bogs are simply colonised by more acid-loving (acidophilic) methanogens. In bogs at least, these methanogens are often related organisms and specialists in this environment. They are variously assigned to the 'Fen', 'R10', 'E1/E2' cluster (the same cluster has various names) within the Methanomicrobiales (Bräuer et al., 2006). Conversely, methanogens have also been isolated from a range of alkaline (high pH) habitats. For example, work Boone et al., in the mid-1980s isolated a range of methanogens from alkaline lakes in Wadi El Natrun, Egpyt, including a strain from Bosa Lake, which has a pH of 9.7 (Boone et al., 1986). Thus, it seems likely that at least in part, any correlation with pH is somewhat artefactual as waters with pH outside of pH 5-8 are somewhat unusual in natural waterbodies.

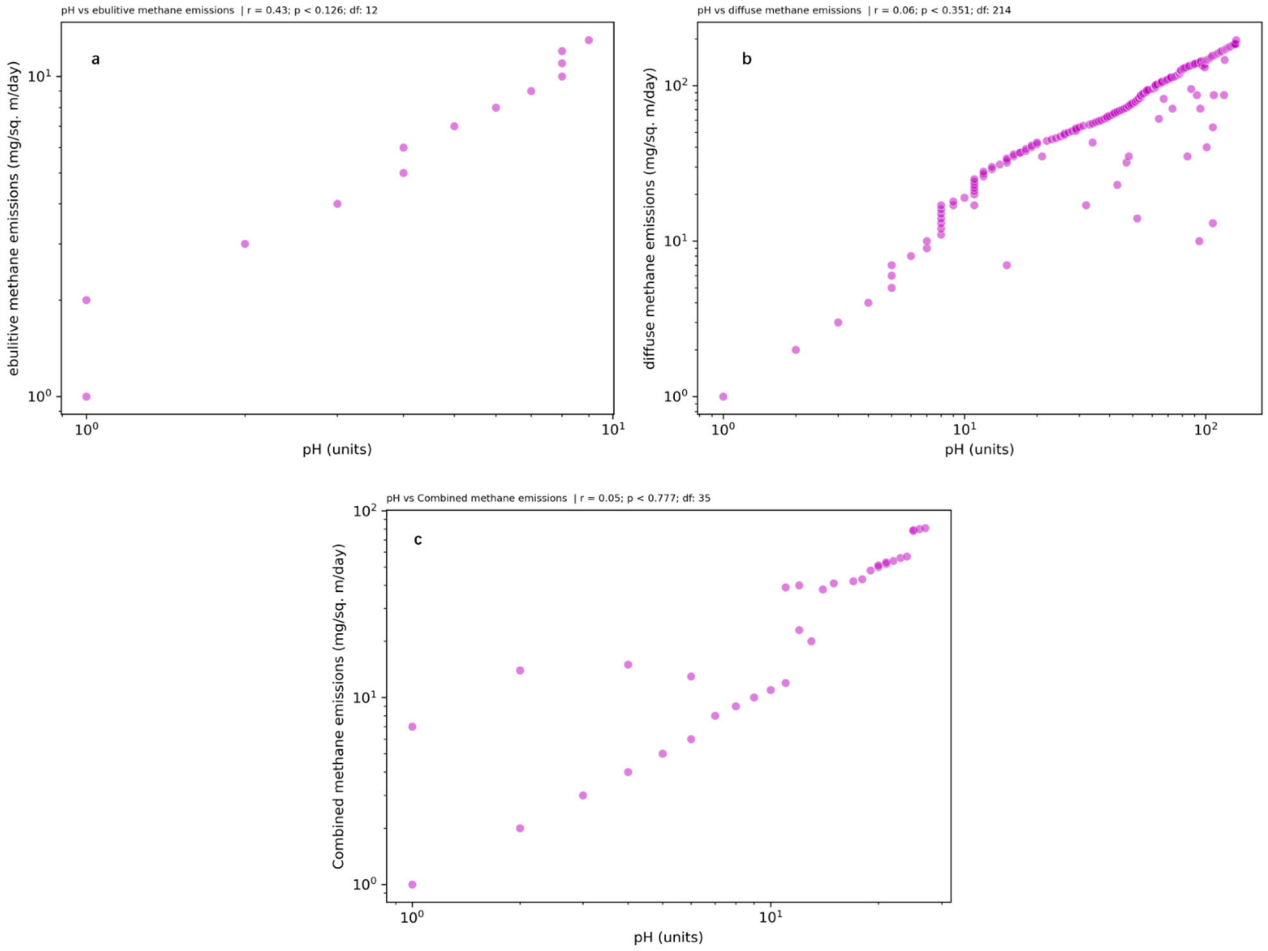


Figure 27 Scatter plots showing relationships between pH and ebullitive emissions (a), diffusive emissions (b) and combined emissions (c). Both X and Y axes of the plots are in logarithmic scales.

5 CSG holding ponds and other aquatic systems in Queensland

5.1 Data collation

Information on CSG holding ponds in Queensland such as pond name, pond dimensions, pond purpose and pond capacity was collated from various publicly available reports from CSG companies (APLNG, 2010; Origin, 2017a, 2017b; QGC, 2012a, 2012b; Santos, 2013, 2012a) and a CSIRO literature review (Mallants et al., 2017). Water chemistry data were obtained through discussions with industry, some industry reports and peer-reviewed papers (AGL, 2014; APLNG, 2010; CRL, 2017; GalileeEnergy, 2022; Origin, 2017b, 2017a; Santos, 2013, 2012b, 2012a; Vick et al., 2019; Webster et al., 2000). In broad terms, the water chemistry data available to the present study was either from produced formation water or brines that result from water treatment. Water chemistry data of brine and produced water in the Bowen and Surat basins were separated for further discussion.

5.2 Data availability

In terms of CSG holding ponds, there are quite a number of ponds without dimension information (e.g., area or depth) except pond volume details (Apx Table C.4). Except the ponds in Apx Table C.4, it is not known whether any other CSG ponds exist based on the current publicly available reports.

Water chemistry data of the CSG holding ponds in the Bowen Basin are from recent analyses, while those in the Surat Basin are from the analyses over the last 10 years. Therefore, more water chemistry data are available in the Surat Basin compared to the Bowen Basin. For the Galilee Basin, water chemistry data of the groundwater and produced water related to coal are only found in two pubic reports (CRL, 2017; GalileeEnergy, 2022). Only limited water chemistry data are available for groundwater associated with the Cooper Basin in Queensland, and what little is available were collected from one industry report (AGL, 2014) and one peer-reviewed paper (Webster et al., 2000). Searches for data related to the Adavale Basin did not reveal any water chemistry data.

5.3 Holding pond information

Key statistics

For publicly available data, details of CSG holding ponds in the Bowen and Surat basins were summarised (see: Apx Table C.4). It should be noted that some holding ponds were in proposal at the time of the reports published and the fate of these ponds since these data were made public is unclear. It is also unclear whether these proposals progressed or did not, and these ponds are not considered in this study. All subsequent analyses presented on water chemistry are based on these data accessed during the course of this study.

To briefly summarise the data, the volumes/ areas of all the CSG ponds were collated or calculated (Table 5). Most ponds were described in the documentation as including produced water or brine ponds, but other descriptors included 'desalination water', 'irrigation water storage', 'oily water discharge pond' 'amended water pond', 'RO cleaning wastewater pond' and 'RO backwash pond' were encountered in the data. Evaporation ponds was restricted in Queensland since 2012 but some of the previous water chemistry data from the evaporation ponds are included in this study. It is noteworthy that much of the information regarding ponds managed by Origin were not publicly available (Apx Table C.4), this means that data in Table 5 underestimates Origin's water holdings.

Taking just the public data, CSG holding ponds for the Surat and Bowen Basin CSG industry hold almost 65,000 ML of water in an area larger than 1100 ha. Violin plots are created to visually display the distribution of pond areas and pond depths (Figure 28) (see Apx B.2 for details of violin plots). The area of individual holding ponds is mostly less than 10 ha with a median size of 7.6 ha (Figure 28a, Apx Table C.4). The calculated pond depths, based on the pond volumes and areas, range from 1.6 to 7.7 m, with most ponds being less than 6 m deep and a median depth of 5 m (Figure 28b).

Table 5 Company, purpose, area and volume of the ponds examined.

Company name	Pond purpose	Total Pond area (ha)	Total Pond volume (ML)
QGC	Brine	93.4	4872
QGC	Desalinated water	9.4	470
QGC	Produced water	389	24453
Santos	Amended water	0.86	43
Santos	Brine	59	2301
Santos	Desalinated water	15	728
Santos	Irrigation water storage	n/a	170
Santos	Produced water	71	3046
Arrow Energy	Brine	60	4000
Arrow Energy	Desalinated water	16	800
Arrow Energy	Oily water discharge	24	1200
Arrow Energy	Produced water	56	2800
Arrow Energy	RO cleaning wastewater	8	400
Arrow Energy	RO backwash	8	400
Origin (APLNG)	Brine	>338	17555
Origin (APLNG)	Produced water	>23.6	1403

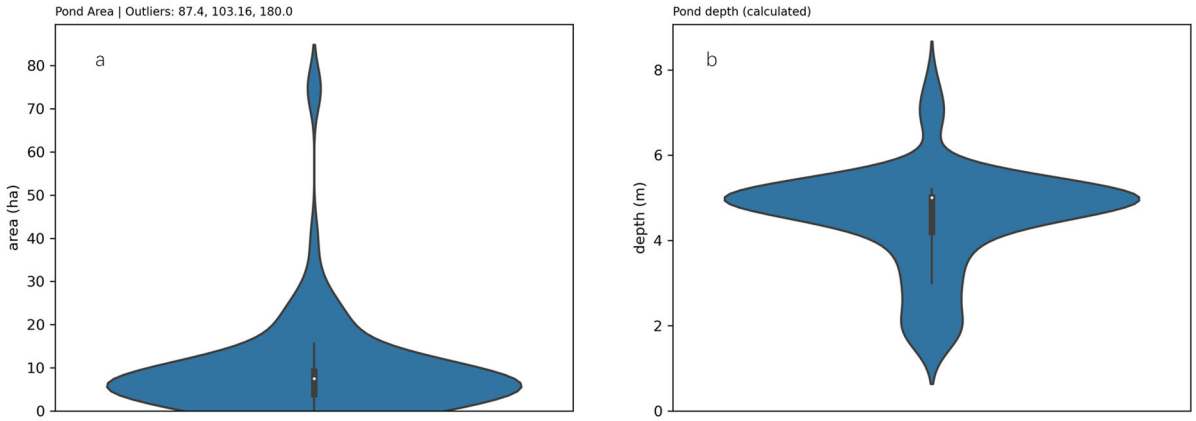


Figure 28 Violin plots showing the distributions of area (a) and depth (b) for CSG holding ponds.

Other than the water holding ponds associated with the CSG industry in the Surat and Bowen basins, the current study did examine other Energy-related water holdings. For example, in the Galilee Basin, five pilot wells, and a monitoring bore were drilled by AGL on Glenaras Station and a produced water holding pond (Glenaras Pond) was constructed for the produced water from the pilot production(GalileeEnergy, 2022). Since 2018, five lateral wells and six vertical wells began production at the Glenaras Multi-lateral pilot, targeting coal seams from the Permian Betts Creek Beds (GalileeEnergy, 2022). Information about water holding ponds for the Glenaras Multi-lateral pilot is not disclosed in the publicly available report. In addition, no water holding pond information is available publicly regarding the Gunn Pilot either (CRL, 2017).

In the Cooper Basin, most of the activities in Queensland are related to conventional oil and gas which produce less water than CSG activity (KCB, 2022). No holding pond information was discovered during this investigation.

In the Adavale Basin, the first and only gas field was discovered is the Gilmore gas field (Troup and Talebi, 2019). No holding pond information was discovered during this investigation.

5.4 CSG holding pond water chemistry

5.4.1 Bowen and Surat basins

Statistical summaries of water chemistry data were collated and are shown in Tables (6, 7, 8, 9, 10 & 11) for the Surat, Bowen and Galilee basins. Most water chemistry data of the CSG holding ponds in the Bowen Basin date from the period 2021-2022, with a few data points from 2012. The water chemistry data of the CSG holding ponds in the Surat Basin were from reports dated 2011 to 2022. Water chemistry data for the Galilee Basin were from the coal seam gas pilot wells at Glenaras pilot and Glenaras Multi-lateral pilot, a monitoring bore at Glenaras pilot and baseline bores at Gunn#2 (CRL, 2017; GalileeEnergy, 2022).

Table 6 Water chemistry of produced water in the Bowen Basin

Analyte (unit)	Limit of detection	Mean	Min.	Median	Max.	Sample count
pH	0.01	8.4	7.4	8.5	11	48
Electrical Conductivity (uS/cm)	1	7892	3701	7470	23282	77
Total dissolved solids (mg/L)	10	5366	1644	4200	13700	9
Bicarbonate as $CaCO_3$ (mg/L)	1	1375	615	1360	2510	26
Carbonate as $CaCO_3$ (mg/L)	1	203	24	204	683	19
Total alkalinity as $CaCO_3$ (mg/L)	1	1524	615	1470	2820	26
Dissolved organic carbon (mg/L)	10	63	11	30	225	21
Total Organic carbon (TOC) (mg/L)	5	62	12	35	243	22
Total nitrogen (mg/L)	0.1	3.4	0.7	3.8	4.9	6
Total phosphorus (mg/L)	0.01	0.38	0.03	0.3	0.96	11
Fluoride (mg/L)	0.1	3.2	0.4	3.9	6.7	23
Chloride (mg/L)	1	1965	664	1520	5680	30
Sulphate as SO_4 (mg/L)	1	2.2	1	2	6	20
Potassium (mg/L)	1	180	4	38	849	30
Sodium (mg/L)	1	1827	852	1760	4840	30

Table 7 Water chemistry of brine in the Bowen Basin

Analyte (unit)	Limit of detection	Mean	Min.	Median	Max.	Sample count
pH	0.01	8.81	8.02	8.94	9.26	15
Electrical Conductivity (uS/cm)	1	55594	34850	54700	80756	15
Total dissolved solids (mg/L)	10	40607	25100	41500	49600	13
Bicarbonate as $CaCO_3$ (mg/L)	1	8643	6280	8295	11300	12
Carbonate as $CaCO_3$ (mg/L)	1	7765	4300	7520	10300	12
Total alkalinity as $CaCO_3$ (mg/L)	1	16400	11300	17350	20700	12
Dissolved organic carbon (mg/L)	10	63	51	63	75	2
Total Organic carbon (TOC) (mg/L)	5	122	74	143	148	3
Total nitrogen (mg/L)	0.1	7.3	5.2	6.4	11	7
Total phosphorus (mg/L)	0.01	4.3	2.03	4.6	5.96	7
Fluoride (mg/L)	0.1	50	15	58	64	7
Chloride (mg/L)	1	16325	9600	15450	27200	12
Sulphate as SO_4 (mg/L)	1	65	47	66	86	12
Potassium (mg/L)	1	347	218	296	566	12
Sodium (mg/L)	1	17567	11200	17400	24100	12

Table 8 Water chemistry of produced water in the Surat Basin

Analyte (unit)	Limit of detection	Mean	Min.	Median	Max.	Sample count
pH	0.01	9	3.5	9	10.4	588
Electrical Conductivity (uS/cm)	1	10399	8	9130	43300	523
Total dissolved solids (mg/L)	10	5090	20	4100	33700	705
Bicarbonate as $CaCO_3$ (mg/L)	1	1399	10	1400	15900	1182
Carbonate as $CaCO_3$ (mg/L)	1	740	9	140	16000	368
Total alkalinity as $CaCO_3$ (mg/L)	1	2091	10	1800	21900	1089
Dissolved organic carbon (mg/L)	10	48	1	14	2400	628
Total Organic carbon (TOC) (mg/L)	5	53	1	15	3200	722
Total nitrogen (mg/L)	0.1	1.9	0.1	1.6	14	546
Total phosphorus (mg/L)	0.01	0.13	0.01	0.08	4.7	635
Fluoride (mg/L)	0.1	4.7	0.1	4.2	48	1059
Chloride (mg/L)	1	1346	3	894	17400	1101
Sulphate as SO_4 (mg/L)	1	22	0.4	7	390	666
Potassium (mg/L)	1	34	0.2	8.2	1010	1129
Sodium (mg/L)	1	1737	7	1400	14500	1139

Table 9 Water chemistry of brine in the Surat Basin

Analyte (unit)	Limit of detection	Mean	Min.	Median	Max.	Sample count
pH	0.01	9.4	4.7	9.4	10.5	281
Electrical Conductivity (uS/cm)	1	53206	4219	42700	187000	195
Total dissolved solids (mg/L)	10	40637	50	17900	280000	188
Bicarbonate as $CaCO_3$ (mg/L)	1	7696	15	6770	99000	400
Carbonate as $CaCO_3$ (mg/L)	1	11529	32	9760	125000	209
Total alkalinity as $CaCO_3$ (mg/L)	1	17547	15	11100	435000	405
Dissolved organic carbon (mg/L)	10	56	3.1	30	461	163
Total Organic carbon (TOC) (mg/L)	5	86	2.7	42	1860	168
Total nitrogen (mg/L)	0.1	6.6	0.38	6.1	35	122
Total phosphorus (mg/L)	0.01	1.3	0.02	0.82	11	182
Fluoride (mg/L)	0.1	45	0.4	27	572	355
Chloride (mg/L)	1	10870	5	7470	77800	364
Sulphate as SO_4 (mg/L)	1	2450	6	2900	6960	193
Potassium (mg/L)	1	103	0.36	29	1410	248
Sodium (mg/L)	1	15413	13	7410	136000	241

Violin plots are created to visually display the distribution of data collected for the various physicochemical parameters of both brines and production waters from the Bowen and Surat basins. It is noteworthy that significantly more data points (up to several hundred times more) were available for the Surat Basin than the Bowen Basin. Due to the significantly higher number of data points available for the produced water and brine in the Surat basin, the distribution of the data for each physicochemical parameters were better sampled compared to those of the Bowen, hence the median values may be more instructive for the Surat Basin. Information pertaining to each chemical parameter discussed below were obtained from Albarède (2009), Andrews et al. (1996), Stumm and Morgan (1996) except other references therein.

Produced water is normally treated using a reverse osmosis plant, which produces very pure water along with a brine that comprises the salts from the produced water (Figure 3). Comparisons revealed a strong correlation (r >0.9) between the solutes in the produced water and in the brine from the same basin, indicating that the brine was derived from reverse osmosis treatment of water of the same chemistry observed in this study (Figure 29).

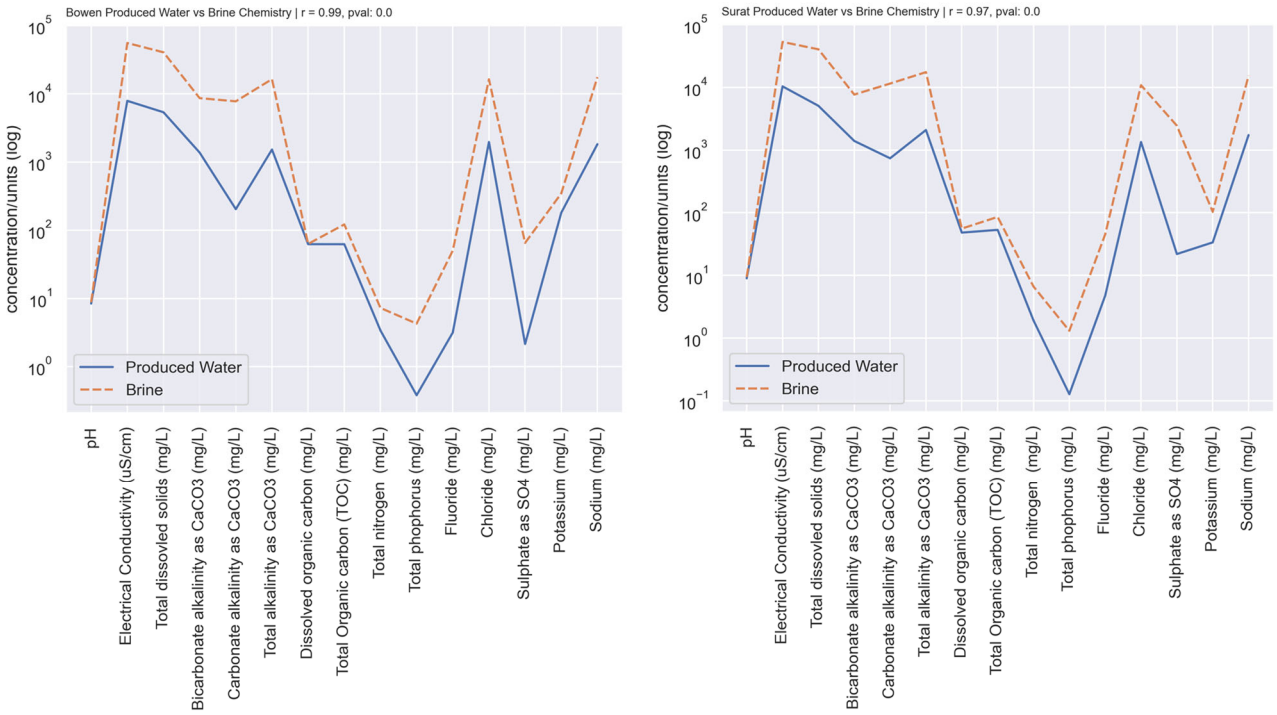


Figure 29 Concentration differences of chemicals in produced water and brine between Bowen (left) and Surat (right) basins

5.4.2 Carbon

Organic carbon

Dissolved organic carbon (DOC) is defined as organic carbon that is capable of passing through a filter (typically between 0.22 and 0.7 µm). Since DOC serves as a food source for aquatic water food webs, it is of primary biogeochemical importance. Furthermore, DOC can affect light penetration, pH (of weakly buffered freshwater systems) and may complex with trace metals. This last is especially true of polyaromatic compounds.

The median DOC values of the Bowen and Surat produced waters are 30 mg/L and 14 mg/L (Tables 6 and 8), respectively, whereas, the Bowen and Surat brines exhibit median DOC values of 63 mg/L and 30 mg/L (Tables 7 and 9), respectively. The fact that DOC values in brines are only doubled compared to the relevant produced water is probably due to its labile character to microbes (Hansell, 2013). The higher DOC content in the produced water and brine from the Bowen Basin compared to the Surat Basin (Figure 30) might be related to the depositional environment of the coal, thermal maturity of the coal, the flux of freshwater into the formation from surrounding formations and the residency time of the water (Robertson, 2018). The different distribution shapes of violin plots between the Bowen and Surat produced water (Figures 30a and 30b) are due to some high DOC values from the evaporation ponds in the Surat Basin.

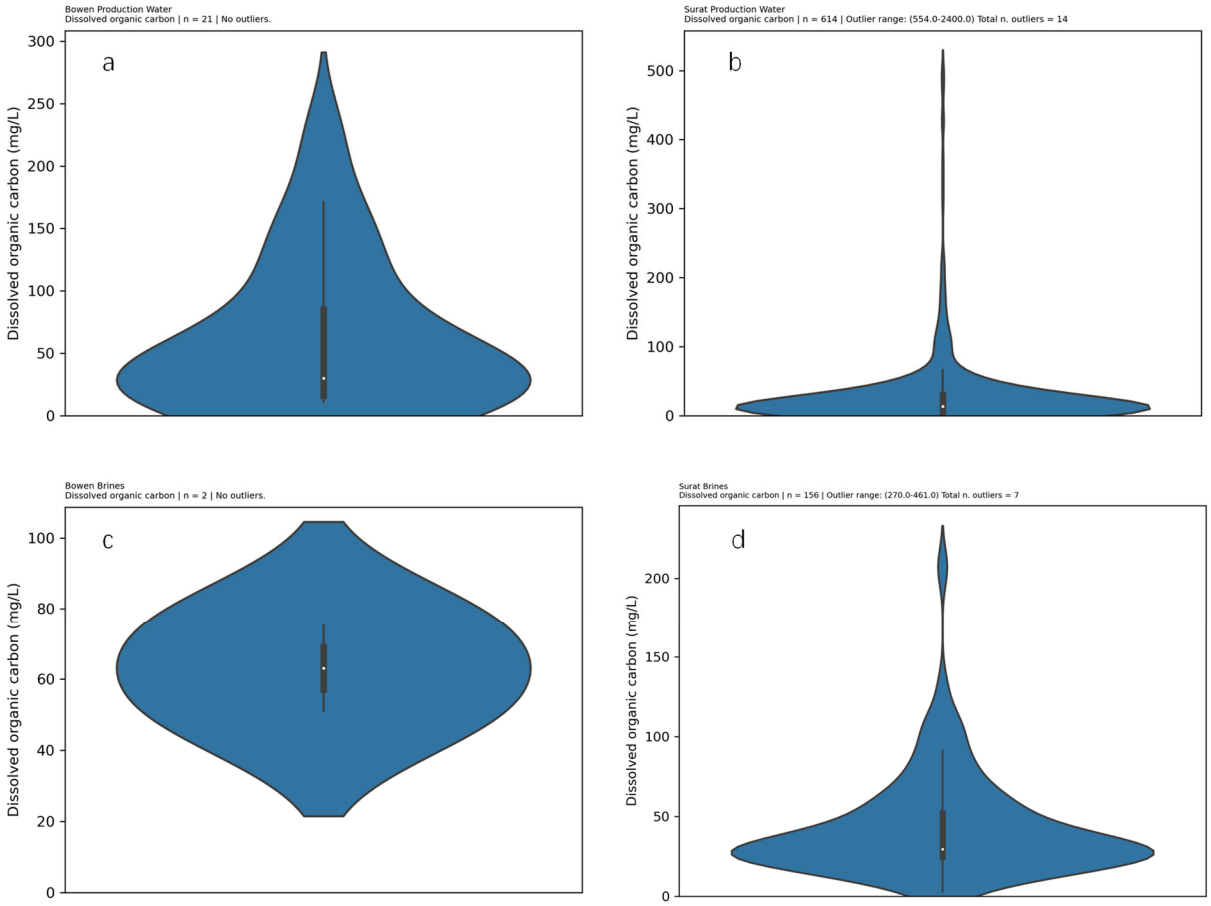


Figure 30 Violin plots showing the distributions of dissolved organic matter in Bowen produced water (a), Surat produced water (b), Bowen brine (c) and Surat brine (d)

Total organic carbon (TOC) is a measure of the total amount of organic carbon in the water. Since all TOC analysers only measure total carbon (TC), TOC analysis always requires some accounting for the inorganic carbon (IC) that is always present. For example, one of the indirect methods is referred to as the TC-TIC method, which measures the amount of IC evolved from an acidified aliquot of a sample and also the amount of TC present in the sample and then TOC is calculated by subtraction of the IC value from the TC of the sample.

TOC provides information on the water quality and organic matter input. The median TOC values of the Bowen and Surat produced waters are 35 mg/L and 15 mg/L (Tables 6 and 8), respectively,

while the Bowen and Surat brines exhibit median TOC values of 143 mg/L and 42 mg/L (Tables 7 and 9), respectively. The TOC in produced water and brine are only slightly higher than the DOC, indicating that most of the organic carbon is present in dissolved form in the CSG ponds. Same as for DOC, the TOC content is higher in produced waters and brines from the Bowen Basin compared to those waters from the Surat Basin (Figure 31). The different distribution shapes of violin plots between the Bowen and Surat produced waters (Figures 31a and 31b) are due to higher TOC values from the evaporation ponds in the Surat Basin.

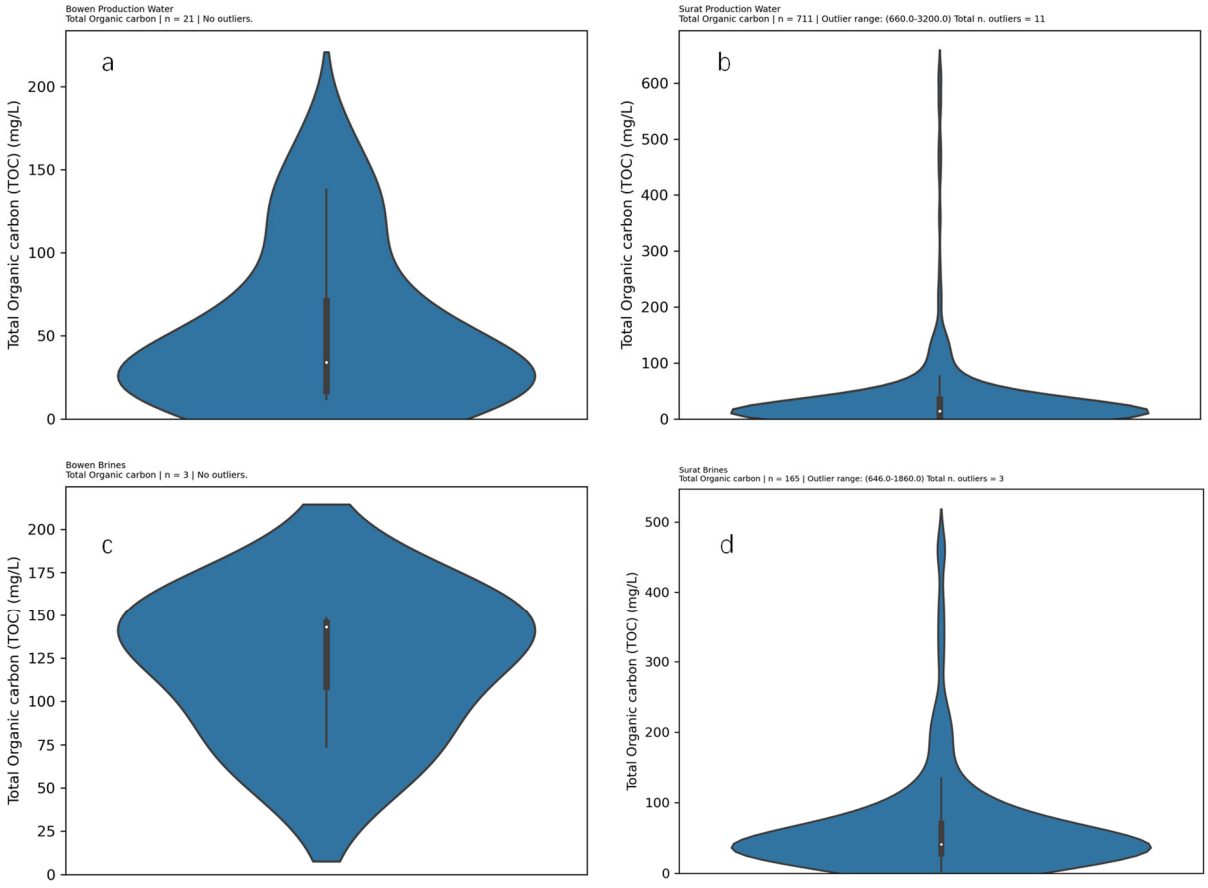


Figure 31 Violin plots showing the distributions of total organic matter in Bowen produced water (a), Surat produced water (b), Bowen brine (c) and Surat brine (d)

It should be noted that these data are probably an underestimate of the true organic carbon content in the ponds, especially produced water ponds which often contain large particles or foreign material such as coal fines, soil, sediment, algae or other materials. These materials settle to the bottom of storage ponds over time and could be a major source of organic carbon for microbes, however this part of organic carbon in the pond is currently unknown. In addition, other post-production inputs may include algal and microbial organic carbon which are derived from chemotrophic or phototrophic processes and use inorganic carbon (bicarbonate from the ponds) as a source of carbon. Both these pools of carbon are very poorly characterised in terms of their concentration and importance.

Inorganic carbon

Inorganic carbon pools in aquatic environments are reliant on pH (Figure 32). The species of inorganic carbon that is present, depends largely on the pH with most of the coal seam waters

examined in the present study being dominated by bicarbonate as they are typically pH 8-9. Chemically, bicarbonate alkalinity is an intermediate form in the deprotonation of carbonic acid to carbonate.

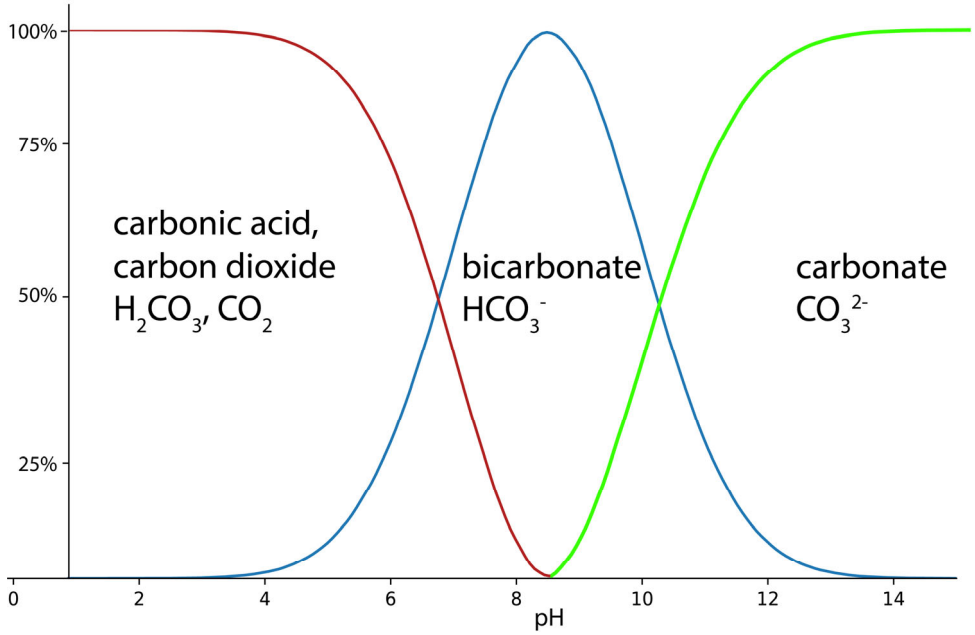


Figure 32 Relationship between pH and inorganic carbon in water

Water in coal seams generally has a high bicarbonate content which can be attributed to two processes: the first process is the dissolution of carbonate by recharge waters (Freeze and Cherry, 1979) either in the coal, or as these waters migrate through other carbonaceous rocks in the subsurface. The second is the biotic and abiotic generation of CO_2 in coal seams which dissolves in water with increasing pressure, and is deprotonated (it loses hydrogen ions) resulting in increased bicarbonate concentration (Taulis and Milke, 2007). The median bicarbonate concentrations of the Bowen and Surat produced waters were 1360 mg/L and 1400 mg/L (Tables 7 and 8). The Bowen and Surat brines exhibit median bicarbonate concentrations of 8295 mg/L and 6770 mg/L (Tables 7 and 9). Even though the median bicarbonate concentrations in produced waters are similar between the Bowen and Surat basins, the distribution of the data are quite different (Figure 33a and 33b), with bicarbonate concentration distributions for the Surat ponds having an almost trimodal distribution. These differences are more pronounced for the brines. Here it can be observed that while most of the Bowen data appears to be drawn from one 'population' of ponds, the Surat brine data show two obvious peaks in the distribution. For instance, in the Surat Basin, there is a group of brines with relatively low bicarbonate concentrations <5000 mg/L and a group of ponds with bicarbonate concentrations above ~6500 mg/L (Figure 33d). This distribution may be the result of some of these ponds having higher pH and more of the bicarbonate being disproportionated to carbonate.

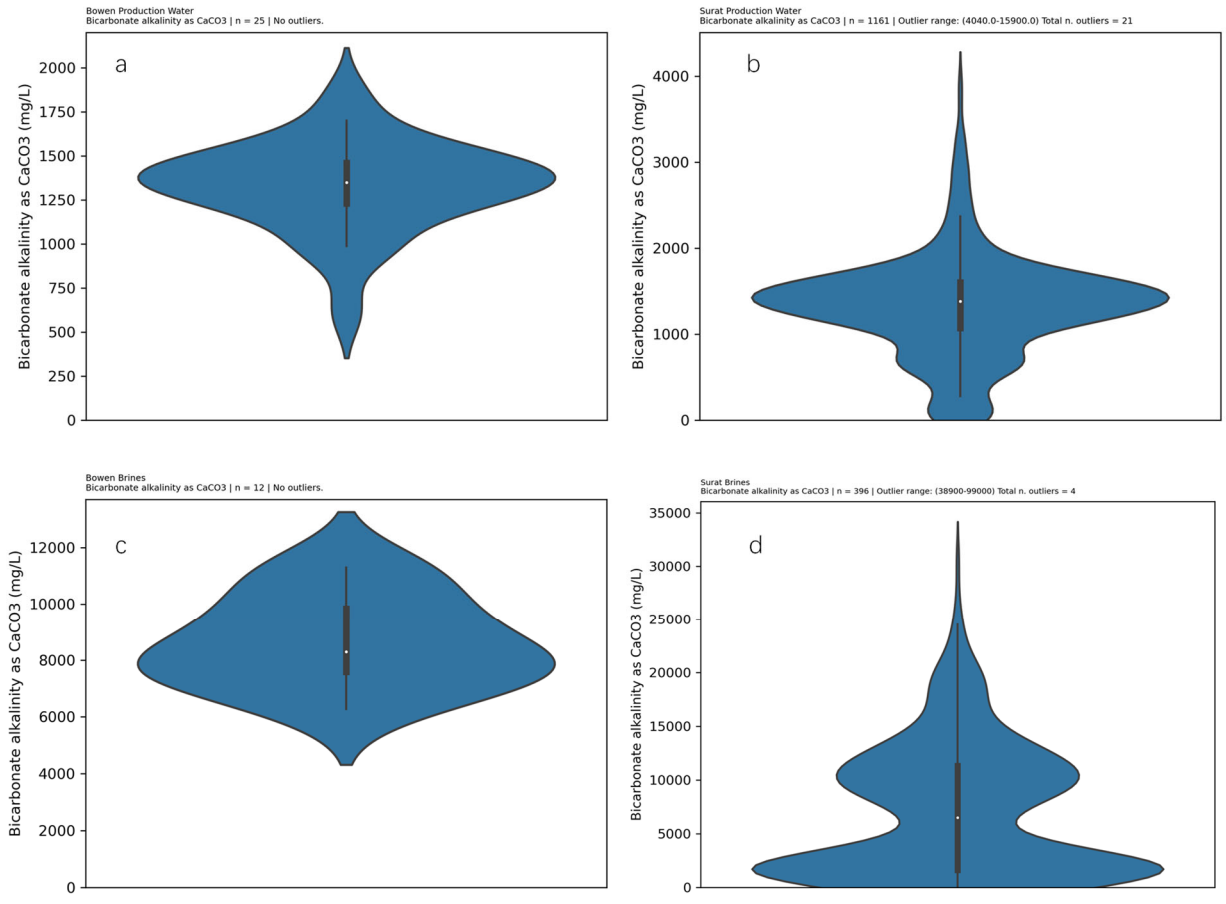


Figure 33 Violin plots showing the distributions of bicarbonate in Bowen produced water (a), Surat produced water (b), Bowen brine (c) and Surat brine (d)

In contrast to bicarbonate (Figure 33a and 33b), the median carbonate concentrations of the Bowen and Surat produced waters were comparatively low (Figure 34a and 34b). Indeed, carbonate concentrations of the Bowen and Surat production waters had only 204 mg/L and 140 mg/L, respectively (Tables 6 and 8), presumably a result of the modestly alkaline conditions of the formation water (Decker et al., 1987). Unsurprisingly, the concentrations of carbonate in Bowen and Surat brines were an order of magnitude higher, with median concentrations of 7520 mg/L and 9760 mg/L, respectively (Tables 7 and 9). Unlike bicarbonate, the distribution of the data for both Bowen and Surat brines and produced waters appeared to indicate that for the most part waters were drawn from the same populations (Figure 34).

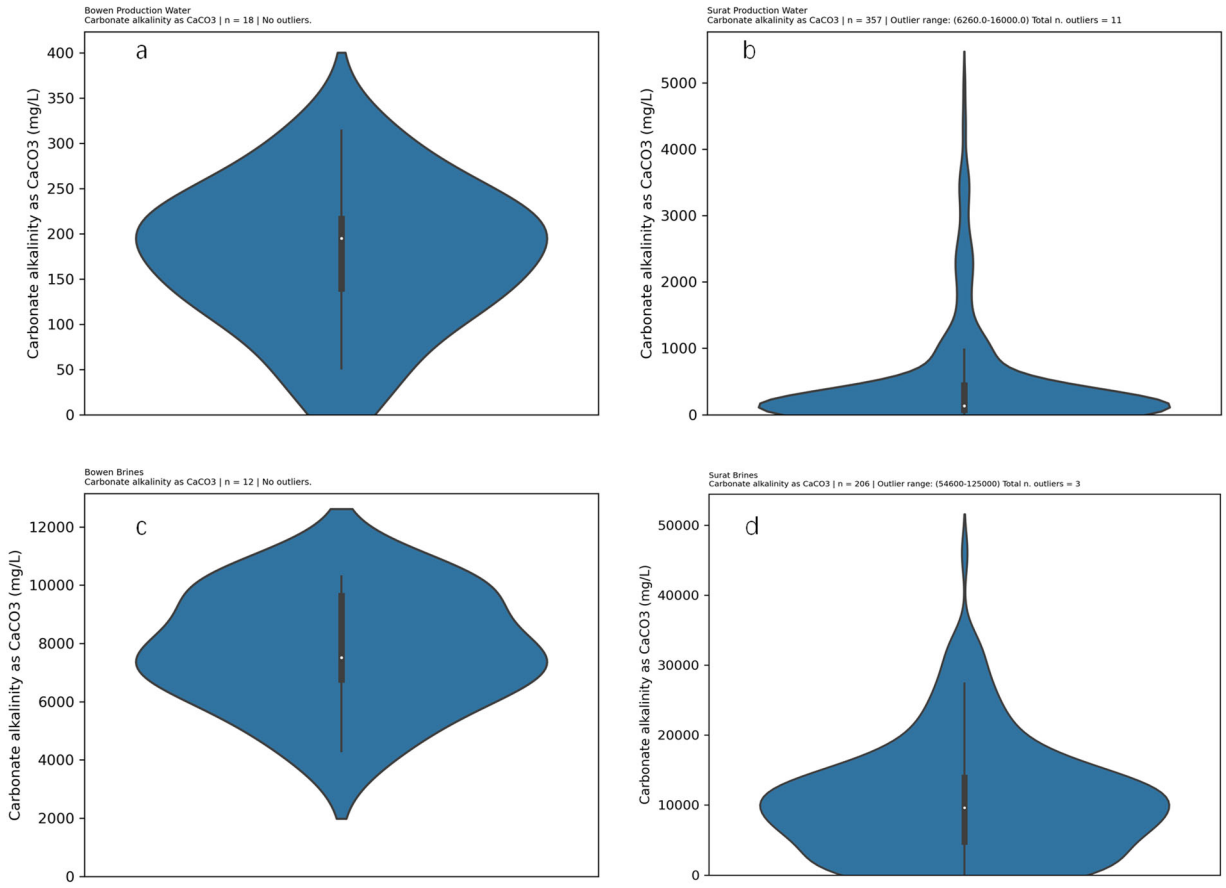


Figure 34 Violin plots showing the distributions of carbonate in Bowen produced water (a), Surat produced water (b), Bowen brine (c) and Surat brine (d)

5.4.3 Total alkalinity

Total alkalinity is reflective of the quantity of all dissolved alkaline chemicals dissolved in water that can both yield and remove hydrogen ions. Total alkalinity as $CaCO_3$ is the sum of carbonate and bicarbonate contents. The median total alkalinity of the Bowen and Surat produced waters were 1470 mg/L and 1800 mg/L, respectively (Tables 6 and 8). The Bowen and Surat brines exhibit median total alkalinity of 17350 mg/L and 11100 mg/L, respectively (Tables 7 and 9). Again, total alkalinity concentrations show a much wider range in the Surat produced water and brine compared to the Bowen produce water and brine, respectively (Figure 35).

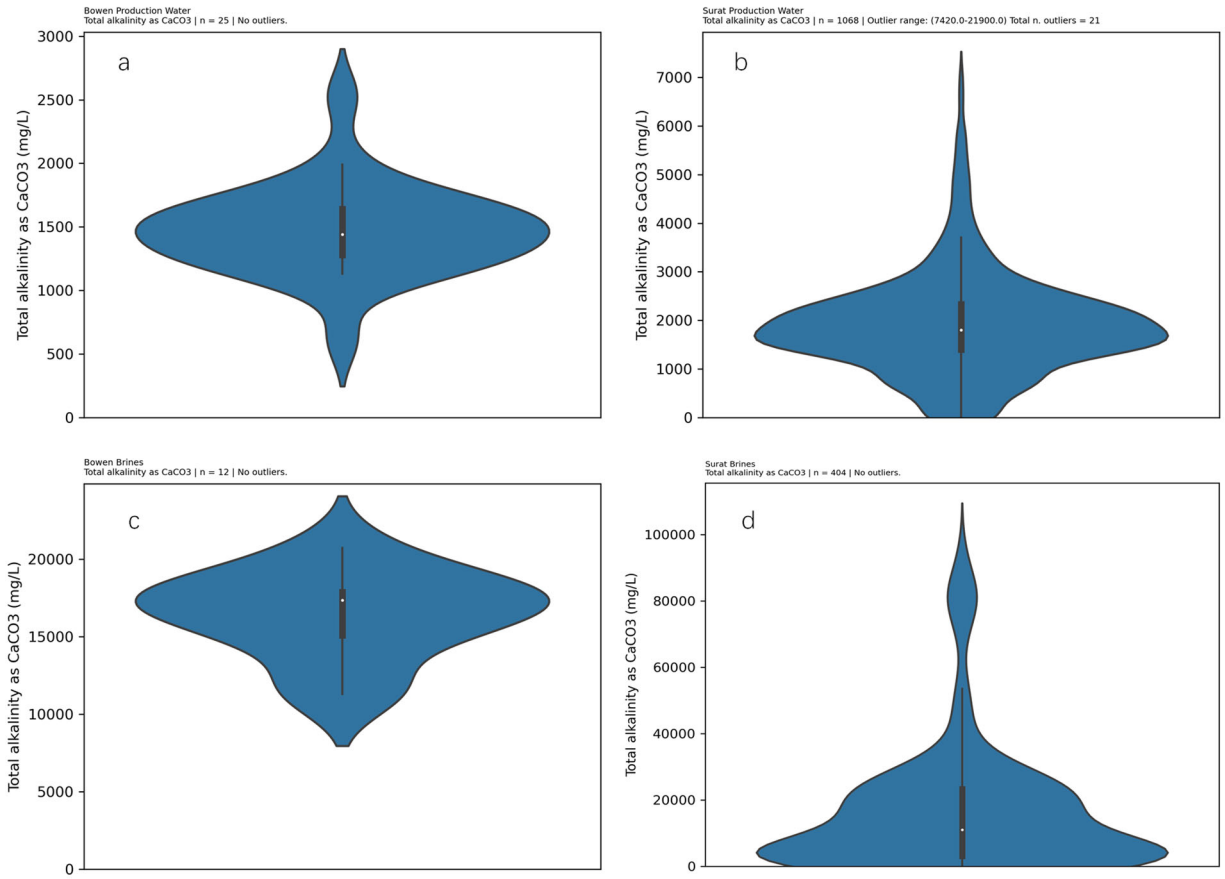


Figure 35 Violin plots showing the distributions of total alkalinity in Bowen produced water (a), Surat produced water (b), Bowen brine (c) and Surat brine (d)

5.4.4 Phosphorus and Nitrogen

The concentration of total phosphorus (TP) is reflective of all phosphorus found in a given sample that encompasses both particulate and dissolved forms. Since phosphorus is a critical nutrient for living organisms, it serves as an important proxy as a limiting nutrient that controls the pace of biological growth. The median TP values of the Bowen and Surat produced waters are 0.3 mg/L and 0.08 mg/L (Tables 6 and 8), respectively. The Bowen and Surat brines exhibit median TP values of 4.55 mg/L and 0.82 mg/L (Tables 7 and 9), respectively. The concentration of phosphorus in general (and across the majority of samples) is considerably lower in the Surat Basin samples compared to samples examined from the Bowen Basin (Figure 36).

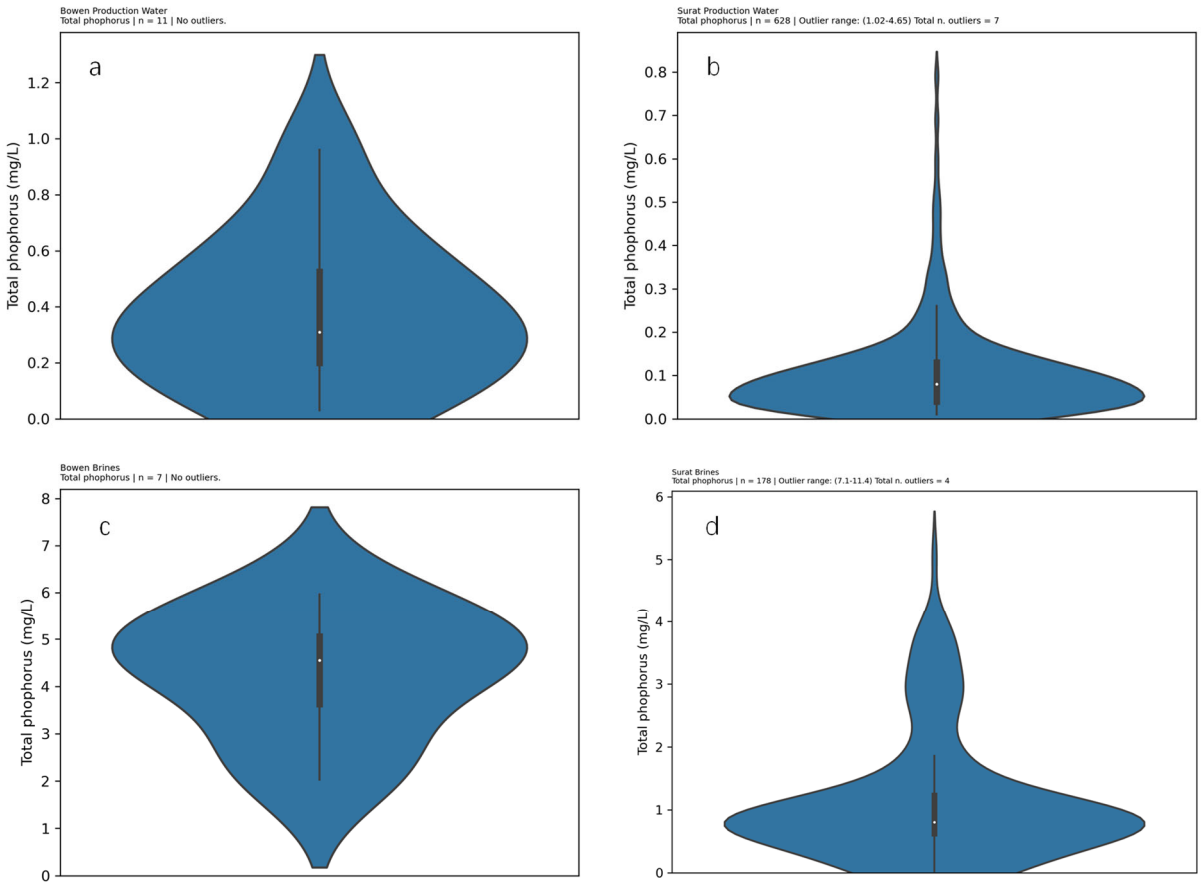


Figure 36 Violin plots showing the distributions of total phosphorous in Bowen produced water (a), Surat produced water (b), Bowen brine (c) and Surat brine (d)

The total nitrogen (TN) reflects the entirety of nitrogen containing compounds. As such it includes organic forms such as amino acids and inorganic forms such as ammonia, nitrite or nitrate. As with phosphorous, nitrogenous compounds are critical for biological growth of organisms in water bodies, but unlike phosphorus are not as frequently limiting. The median TN concentrations of the Bowen and Surat produced waters were 3.8 mg/L and 1.6 mg/L, respectively (Tables 6 and 8). The Bowen and Surat brines exhibit median TN concentrations of 6.4 mg/L and 6.1 mg/L, respectively (Tables 7 and 9). Similar to TP, the median value of TN in the Surat produced waters is nearly half of that observed in Bowen produced waters. Most of the TN concentrations in the Bowen produced waters are in a range of 3-5 mg/L while the TN concentrations in the Surat produced waters are mainly distributed between 1-2 mg/L (Figure 37a and 37b). The TN concentration distribution between the Bowen and Surat brines is similar except some higher values (>15 mg/L) were observed in the Surat brines (Figures 37c and 37d).

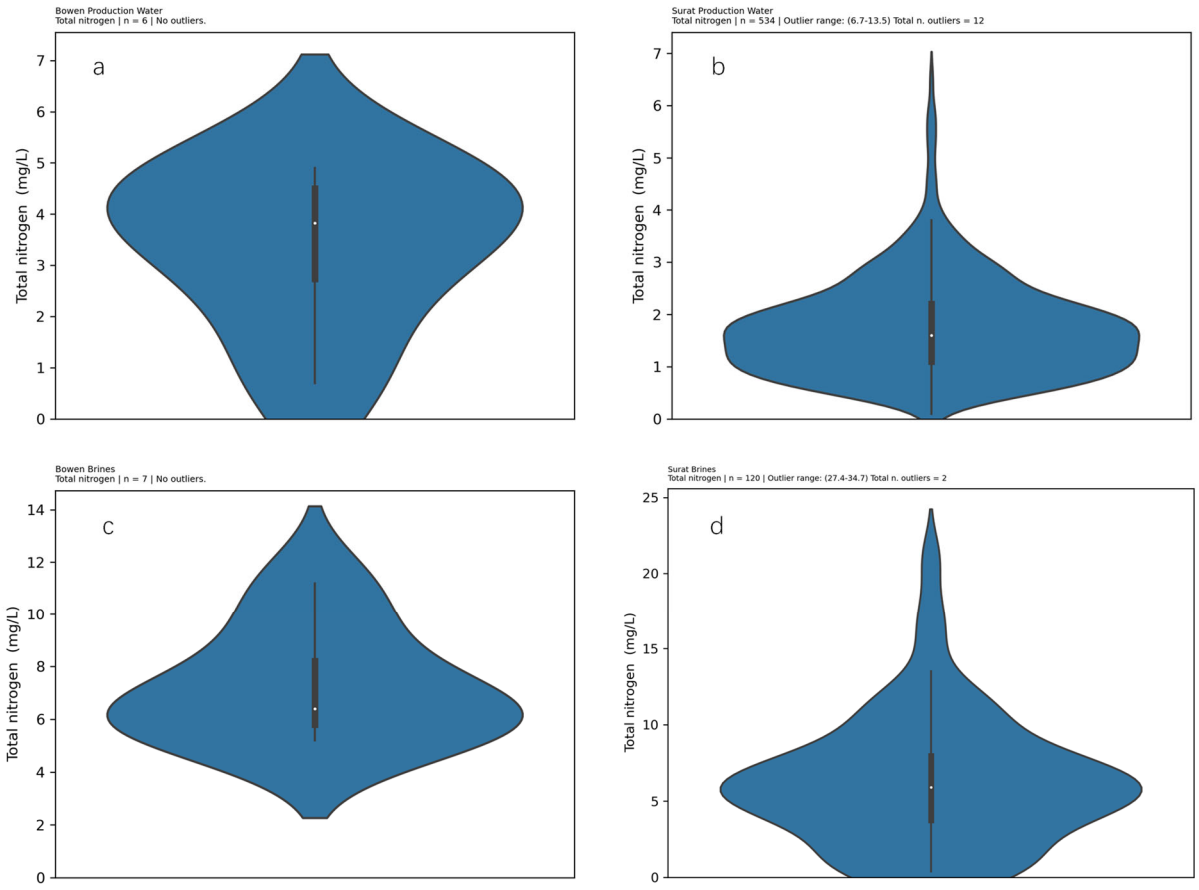


Figure 37 Violin plots showing the distributions of total nitrogen in Bowen produced water (a), Surat produced water (b), Bowen brine (c) and Surat brine (d)

5.4.5 pH

The pH is a measure of the acidity or basicity of a solution (including water in the ponds examined in the present study). pH values less than 7 indicate the solution is acidic, while pH values above 7 indicate the solution is basic. It is important for readers to note that the pH scale is logarithmic, that is pH 6 is ten times more acidic than pH 7, while pH 5 is 100x more acidic than pH 7. pH is a particularly important parameter as it impacts both chemical and biological aspects of the pond substantially. The median pH values of the Bowen and Surat produced waters were 8.5 and 9.1, respectively (Tables 6 and 8). The pH of the Bowen and Surat brines have median values of 8.9 and 9.4 (Tables 7 and 9), respectively. The pH values of produced waters are similar across the Bowen and Surat basins (Figure 38).

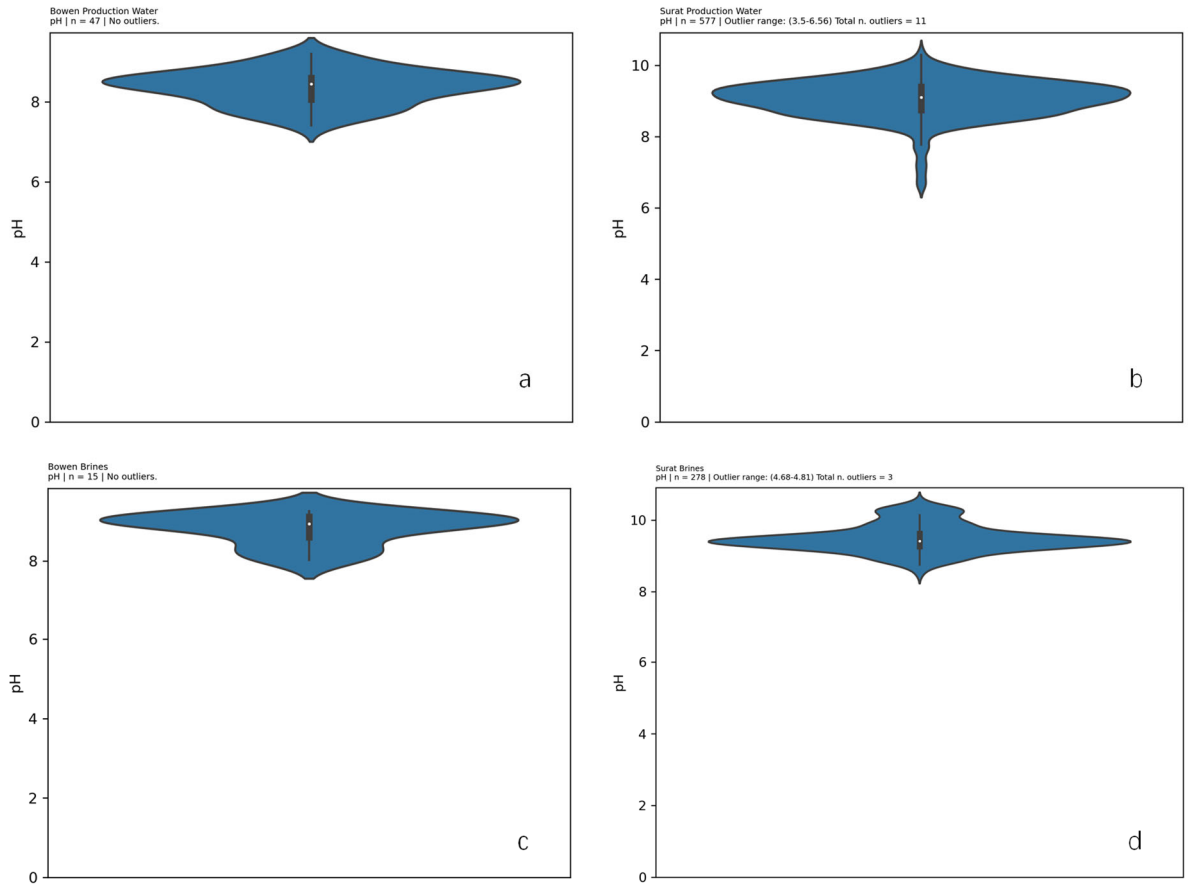


Figure 38 Violin plots showing the distributions of pH in Bowen produced water (a), Surat produced water (b), Bowen brine (c) and Surat brine (d)

5.4.6 Salinity

Salinity is a generic term for 'saltiness' with waters with higher salinity waters having more dissolved salts. The salinity of a water body is a key control on the kinds of microbes that occur in that water and thus has major impacts on microbial diversity in aquatic environments. These dissolved salts can take many forms, for example, sodium chloride or potassium chloride or magnesium sulfate dissolved in water would add to the 'salinity'. Taking just sodium and chloride for example, the median sodium and chloride concentrations of the Bowen produced water were 1760 mg/L and 1520 mg/L (Table 6). For the Surat produced water, the median sodium and chloride concentrations are 1400 mg/L and 894 mg/L (Table 8). The Bowen brines exhibit median sodium and chloride values of 17400 mg/L and 15450 mg/L (Table 7), while the Surat brines have median sodium and chloride concentrations of 7410 mg/L and 7470 mg/L (Table 9). To put this numbers in context, the sodium and chloride concentrations in seawater are around 10800 mg/L and 19300 mg/L (Millero, 1974).

The sodium and chloride concentrations of the Surat produced water are lower than those of the Bowen produced water (Figures 39a and 39b and Figures 40a and 40b). The sodium and chloride concentrations of the Surat brines are much lower than those of the Bowen brines (Figures 39c and 39d and Figures 40c and 40d) with some values even above 30000 mg/L.

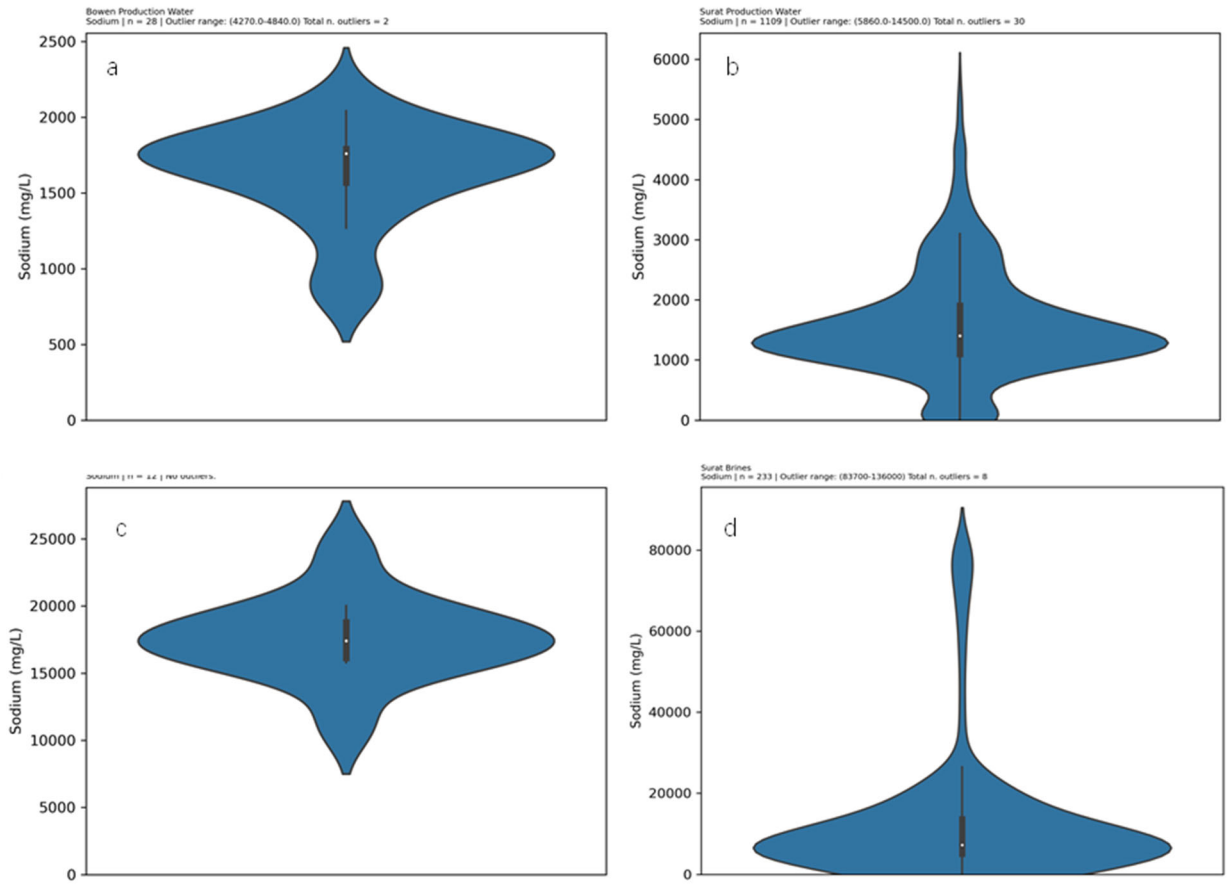


Figure 39 Violin plots showing the distributions of sodium in Bowen produced water (a), Surat produced water (b), Bowen brine (c) and Surat brine (d)

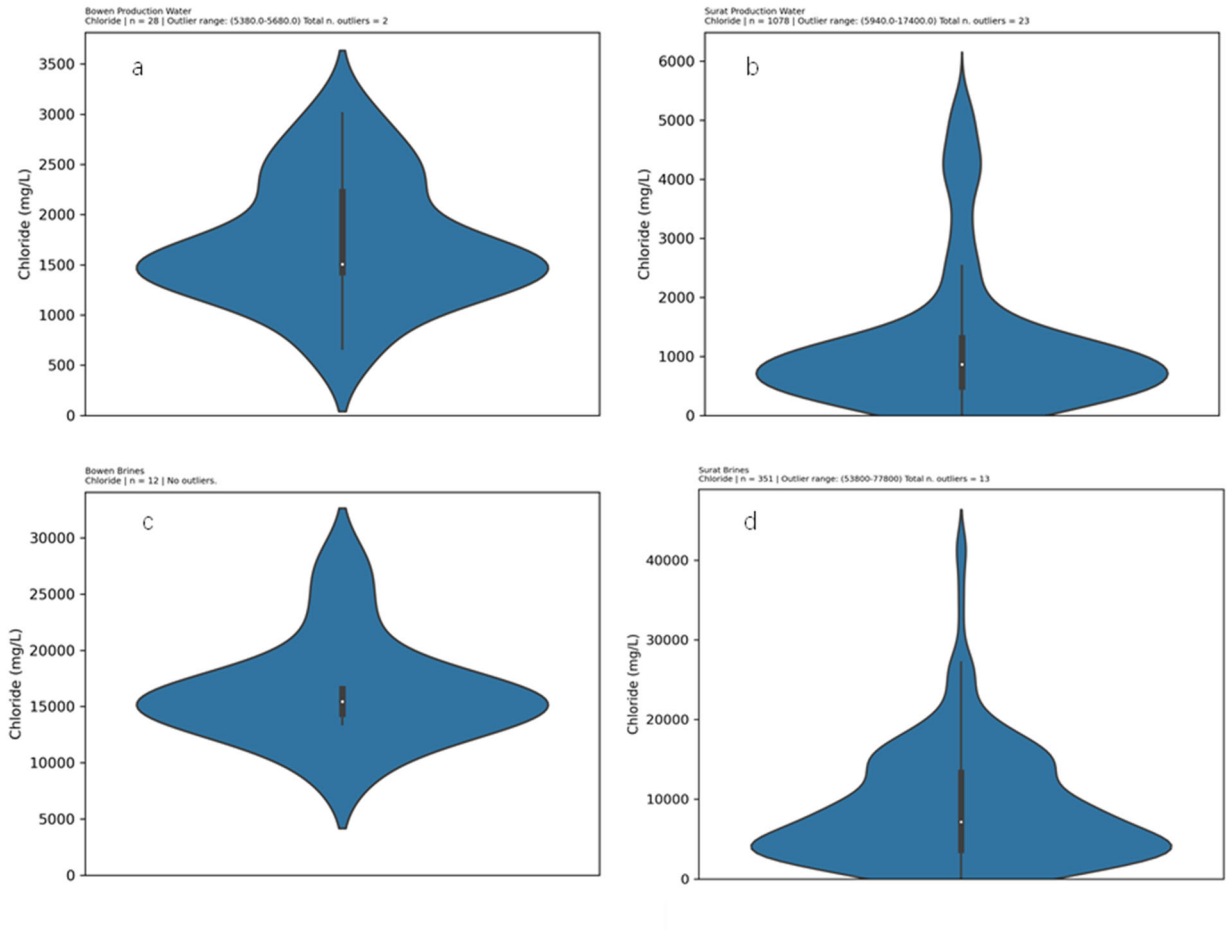


Figure 40 Violin plots showing the distributions of chloride in Bowen produced water (a), Surat produced water (b), Bowen brine (c) and Surat brine (d)

Electrical conductivity (EC) is a measure of the saltiness of a water body and is measured on a scale from 0 to 50,000 uS/cm. The measurement is based on how easily a current moves through water and thus is a measure of the charged ions (and other chemical species) that occur in a liquid. For reference, typical sea water has a conductivity value about 54,000 uS/cm (Turner and Acworth, 2004). The median EC of the Bowen and Surat produced waters were 7470 mg/L and 9130 mg/L, respectively (Tables 6 and 8). The median EC of the Bowen and Surat brines were 54700 mg/L and 42700 mg/L (Tables 7 and 9), respectively. The EC distributions between the Bowen and Surat produced waters are different with a narrow range of 6000-8000 mg/L for the Bowen produced water and a wide range of 3000-15000 mg/L for the Surat produced water (Figures 41a and 41b). The EC distributions between the Bowen and Surat brines were similar except that the Surat brines feature some ponds with very extreme high (>100000 mg/L) or low (20000 mg/L) ECs (Figures 41c and 41d).

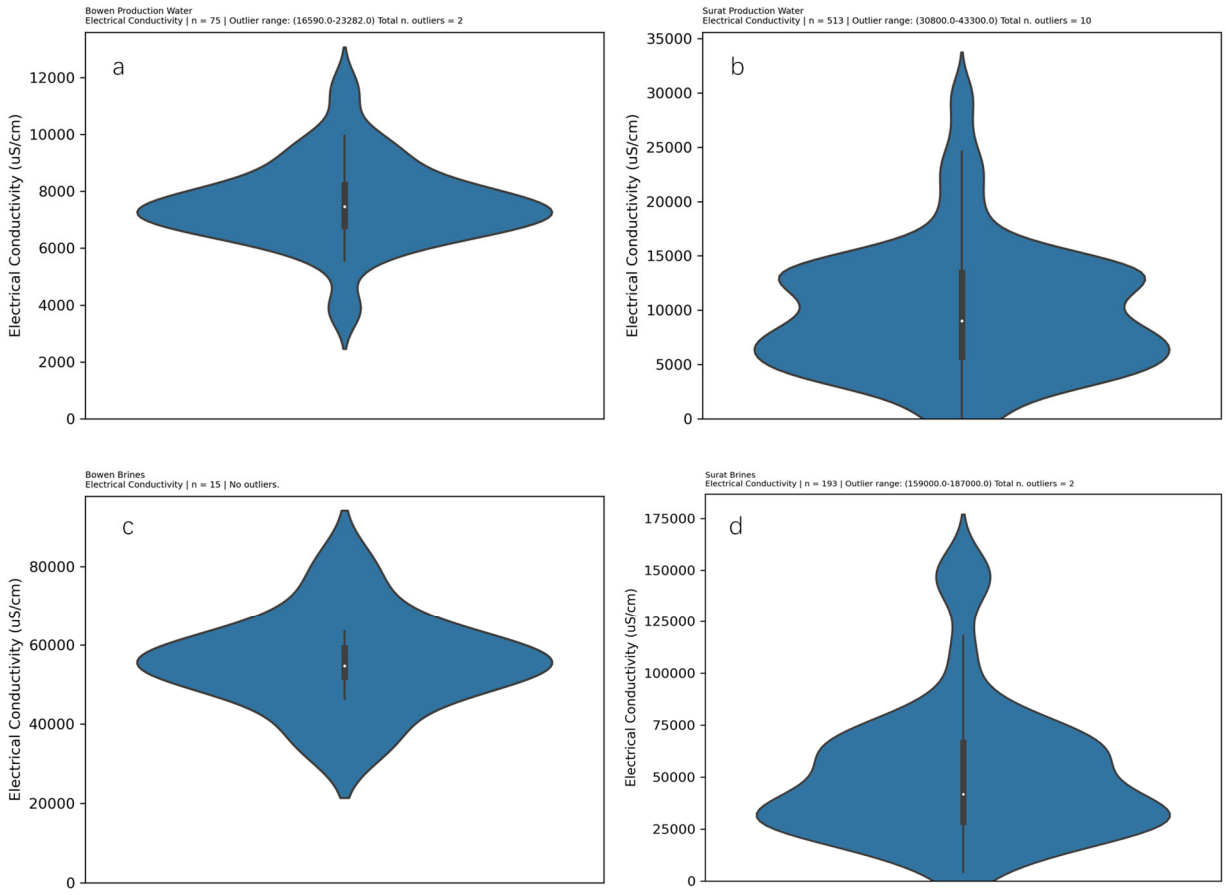


Figure 41 Violin plots showing the distributions of electrical conductivity in Bowen produced water (a), Surat produced water (b), Bowen brine (c) and Surat brine (d)

Total dissolved solids (TDS) encompass the entirety of dissolved inorganic and organic solids present in a liquid less than 2 microns, and unlike EC, TDS includes nonconductive components of the water such as dissolved organic matter. TDS in coal seam produced waters likely include a host of metal salts, carbonates, along with various organic materials that include nitrogen and carbon (Millar et al., 2016). The median TDS concentrations of the Bowen and Surat produced waters were 4200 mg/L and 4100 mg/L, respectively (Tables 6 and 8). The median TDS of the Bowen and Surat brines were 41500 mg/L and 17900 mg/L (Tables 7 and 9). respectively. The distribution of TDS across the ponds varied between the Bowen and Surat produced waters (Figure 42a and 42b). The median TDS of the Bowen brines is two times higher than that of the Surat brines (Figures 42c and 42d), and likely reflective of differences in sodium and chloride concentrations.

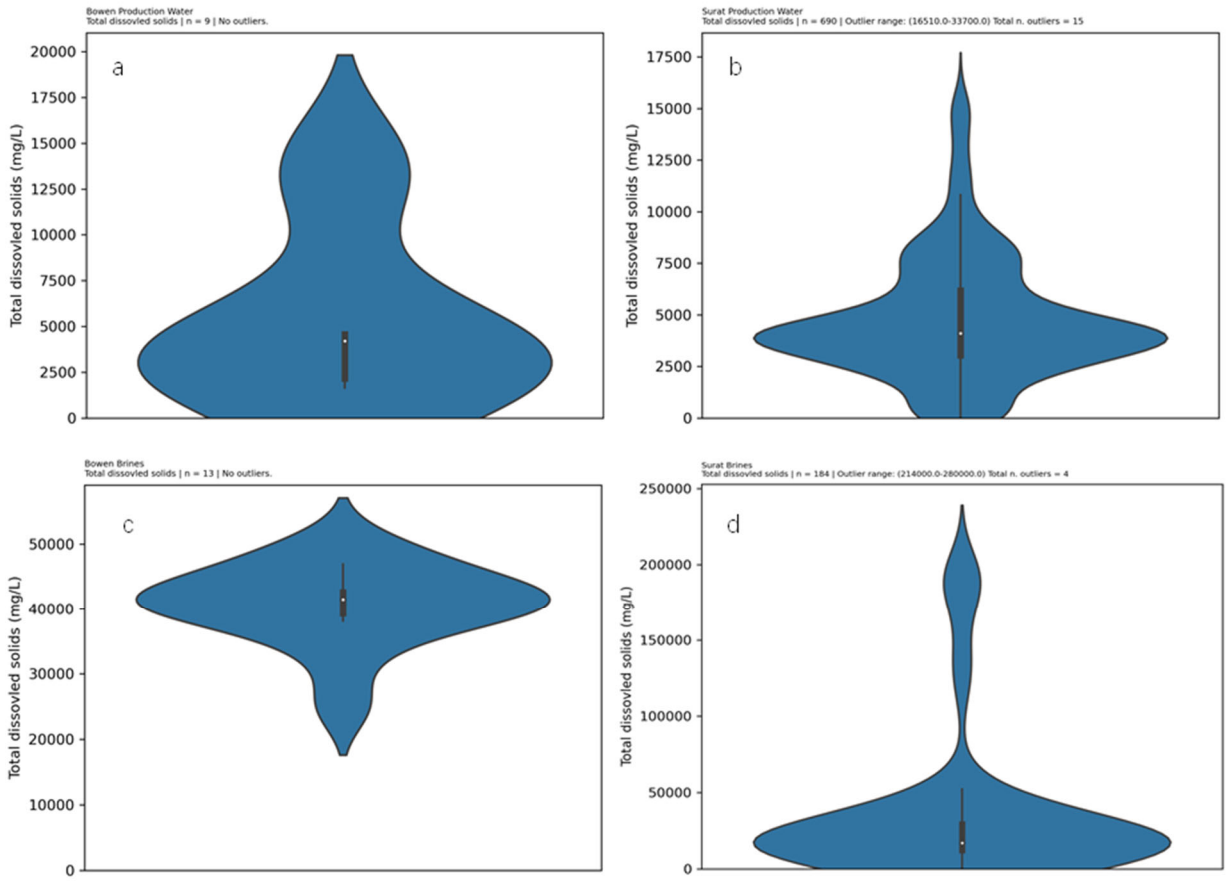


Figure 42 Violin plots showing the distributions of total dissolved solids in Bowen produced water (a), Surat produced water (b), Bowen brine (c) and Surat brine (d)

5.4.7 Sulphate

Sulphate in water represents an important macronutrient for microbial activity. This compound is important as a source of sulfur for key amino acids (cysteine and methionine) which is vital for some proteins, and additionally the compound can act as an electron acceptor in anoxic environments (where it is reduced to sulfide). In aquifers in coal seams, sulphate increases when fresh recharge water encounters, and dissolves, sulphate minerals such as gypsum and anhydrite, along the path of flow or through the weathering and oxidation of pyrite (Taulis and Milke, 2007), though this last typically cannot occur in anoxic settings. The amount of sulphate available for reduction can also be an important determinant of methanogenesis rates (Watson and Nedwell, 1998). Median sulphate concentrations of the Bowen and Surat produced waters were 2 mg/L and 7 mg/L, respectively (Tables 6 & 8), while Bowen and Surat brines exhibit median sulphate concentrations of 65.5 mg/L (Table 7) and 2900 mg/L (Tables 7 and 9), respectively. The Surat brines contain extremely high concentrations of sulphate (Figure 43).

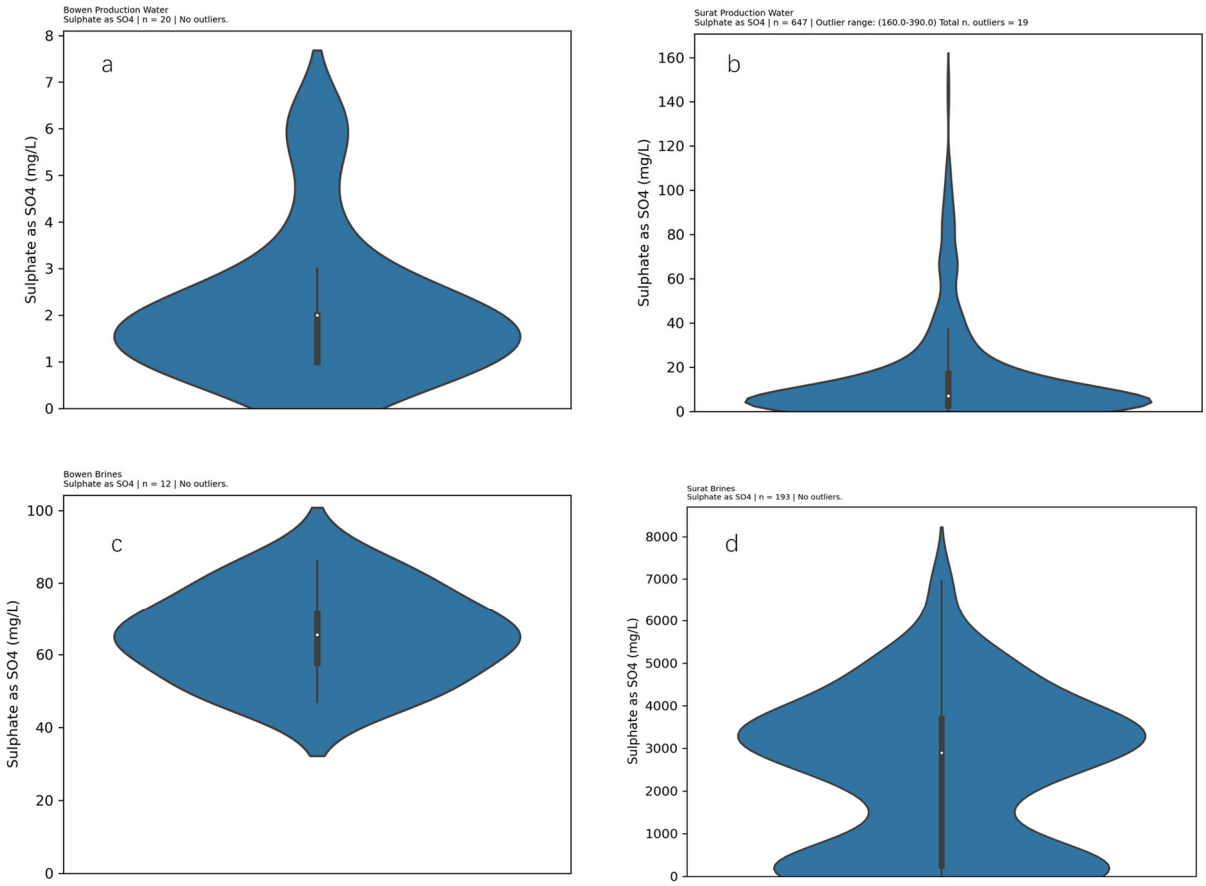


Figure 43 Violin plots showing the distributions of sulphate in Bowen produced water (a), Surat produced water (b), Bowen brine (c) and Surat brine (d)

In summary, based on the median concentration, the Surat produced waters have lower organic carbon, carbonates, nutrients and salinity, and higher total alkalinity, EC and sulphate. similar bicarbonate, TDS and pH are similar to the Bowen produced waters. Surat brine has lower organic carbon, bicarbonate, total alkalinity, nutrients, TDS, EC and salinity, but higher carbonate and sulphate levels. The pH is similar to the Bowen brines. It is noteworthy that more water chemistry data would be useful for the Bowen Basin, as the area is not as well samples in the dataset examined here.

5.4.8 Galilee Basin

Due to the limited activity in the Galilee basin relative to the Bowen and Surat, the data are presented separately for this location. Water chemistry data for the Galilee Basin was from the coal seam gas pilot wells at the Glenaras pilot and Glenaras Multi-lateral pilot, a monitoring bore at the Glenaras pilot and baseline bores at Gunn#2 (Table 9, CRL, 2017; GalileeEnergy, 2022). The statistical summaries are shown in Tables 10 & 11.

Table 10 Water quality data of produced waters from pilot wells in the Galilee Basin

Analyte (unit)	Unit	Limit of detection	Min.	Max.	Median	Sample number
pH	pH unit	0.01	6.99	8.38	7.26	10

Electrical Conductivity	uS/cm	1	1700	2320	1920	33
Total dissolved solids	mg/L	10	915	1270	1090	26
Total alkalinity as $CaCO_3$	mg/L	1	622	846	696	33
Sodium (mg/L)	mg/L	0.1	372	484	434	33
Chloride (mg/L)	mg/L	1	97	425	222	33
Potassium (mg/L)	mg/L	1	9	40	18	33
Sulphate as SO_4 (mg/L)	mg/L	1	1	21	6.5	10

Table 11 Water quality data of ground water from a monitoring well and a baseline well in the Galilee Basin

Analyte (unit)	Unit	Limit of detection	Min.	Max.	Median	Sample number
pH	pH unit	0.01	7.7	8.38	8.2	5
Electrical Conductivity	uS/cm	1	767	1780	1715	6
Total dissolved solids	mg/L	10	452	1080	1030	5
Total alkalinity as $CaCO_3$	mg/L	1	315	846	758	6
Sodium (mg/L)	mg/L	0.1	183	484	438	6
Chloride (mg/L)	mg/L	1	46	126	98	6
Potassium (mg/L)	mg/L	1	3	28	12	6
Sulphate as SO_4 (mg/L)	mg/L	1	n/a	n/a	n/a	n/a

For the Galilee Basin, all formation water pH values range from 7 to 8.4. The EC, TDS, total alkalinity, sodium values are quite similar between the water from the pilot wells and the monitoring well. But the water from the monitoring bore and the baseline bore data indicate lower concentrations of chloride than the pilot wells (Tables 10 & 11). There is no sulphate detected in the monitoring bore and the baseline bores. The water appears to be much fresher than the produced water in the Bowen and Surat basins.

5.5 Other aquatic water systems in Queensland

As stated at the outset, water chemistry data related to energy resource exploration from other basins is very limited. Herein we focus on publicly available reports and peer-reviewed journal articles related to the Cooper Basin. No relevant water chemistry data was found for the Adavale Basin.

To our knowledge, no publicly available water chemistry data in the Cooper Basin in Queensland are related to coal seams. There is only very limited water chemistry data related to conventional oil and gas reservoirs (AGL, 2014; Webster et al., 2000). Water samples collected from bores tapping the Hutton Sandstone (main target for oil production) showed the groundwater is slightly alkaline (pH 7.89), slightly saline (~ 6300 uS/cm), and is dominated by sodium-potassium-chloride-bicarbonate (Na-K-Cl-HCO_3) ions (UWIR, 2014). The water samples collected from bores tapping the Namur Sandstone (target for oil production) showed that the ground water is slightly alkaline (pH 8.2), slightly saline (2250-4800 uS/cm), and is dominated by sodium-bicarbonate (Na-CO_3) ions (AGL,

2014). A recent report provided some water chemistry information on the Cooper Basin in the southwestern Queensland region (KCB, 2022), in which, the water types were described as either sodium-bicarbonate or sodium-bicarbonate-chloride. The total dissolved solids reported in the Hutton Sandstone range from 782 to 5523 mg/L while the groundwater from the major gas reservoir hosted formations has TDS values range from 1920-19243 mg/L (Webster et al., 2000).

5.6 Microbiology related to CSG holding ponds

A literature review on the microbiology of coal seam formation water holding ponds reveals very little data available in published papers. Indeed, in Australia there appears to be no microbial community data available for CSG holding ponds. There are, however, some limited data available from culture-based studies. See Box 1. for a detailed explanation of the differences between microbial community profiling and a culture-based assessment of which organisms occur in a given environment.

In particular, there are two notable studies that warrant some exploration. The first is Bos (2021). In her thesis, Bos explores culturable microbes from coal seam formation water, and brine holding ponds in Queensland and NSW. Data presented in the thesis revealed that two Pseudomonas species (*P. mendocina* and *P. guguanensis*) were identified from coal seam formation water holding ponds. In contrast, from brine samples two isolates were obtained, one a strain of *Dietzia natronolimnaea* and the other a strain of *Bacillus velezensis*.

In addition to the work of Bos (2021), some sequences from cultures obtained from coal seam formation water ponds are available in Genbank (Table 12). These sequences were generated from cultures obtained from various parts of the coal seam gas water infrastructure near Roma, Queensland. Most of these isolates are represented by unpublished (in the literature sequences either available through GenBank or IMG databases.

Table 12 Culture obtained from CSG holding ponds

Culture	Collected from	Reference
Pseudomonas mendocina	Coal seam formation water holding ponds	Bos (2021)
Pseudomonas guguanensis	Coal seam formation water holding ponds	Bos (2021)
Dietzia natronolimnaea	Brines	Bos (2021)
Bacillus velezensis	Brines	Bos (2021)
Anoxybacillus sp. P3H1B	Isolated from coal seam gas collection system.	JGI IMG Taxon ID: lqlnpqskjq
Cellulosilyticum sp. I15G10I2	Isolated from a coal seam gas water treatment pond at the Spring Gully water treatment facility	Adelskov and Patel, 2017
Hyphomicrobium sp. strain NDB2Meth4	Spring Gully water treatment facility, Roma, Queensland	BioProject PRJEB17664
Pannonibacter sp. I15F10I1	Spring Gully water treatment facility, Roma, Queensland	BioProject PRJEB20968
Pannonibacter sp. P2PFMT1	Spring Gully water treatment facility, Roma, Queensland	BioProject

		PRJEB20970
Stappia sp. P2PMeth1	Spring Gully water treatment facility, Roma, Queensland	BioProject PRJEB20986
Stappia sp. TSB10P1A	Spring Gully water treatment facility, Roma, Queensland	BioProject PRJEB20985

Regardless, this study reveals that microbiology of both the coal seam holding ponds and brines remains a significant knowledge gap. The availability of isolates, while useful for a range of studies, do not provide significant information on the microbiological diversity (or even which species are common) in either CSG produced water pools or the resultant brines.

Box 1 Culture based studies vs. microbial community profiling

It is worth noting that there is a marked difference between which organisms occur in a given environment, and which organisms can be cultured (i.e., grown in laboratory) from that environment.

In order to determine which microbes occur in a given environment, the tool that is generally used is genomics. Either sequencing the environmental DNA (eDNA) from an environment (for water this involves filtering out the organisms and extracting the DNA from this filtrate) resulting in what is called metagenomic data. A metagenome is simply the result of sequencing the extracted DNA without focussing on a specific gene or genetic region. This yields mostly DNA from the more common organisms in the environment. The other approach (variously called ecogenomics, ecogenetics, 16S (or other gene) community surveys overcomes this limitation by using a process called PCR to amplify a specific gene. These latter approaches allow microbiologists to identify most of the organisms in a community but does not provide genetic information beyond the identity of the organism in question.

In contrast to these so-called “culture-independent” approaches, culture-based approaches are reliant on particularly microbes growing in a laboratory in order to identify these organisms. For example, a microbiologist might use ‘nutrient agar’ to grow microbes from a coal seam formation water holding pond, however, the microbes that grow may not be common in the original pond water, instead, they are likely just well adapted for growth on particular media like nutrient agar. Further exacerbating this difference is that many microbes are difficult to grow in culture. One upside, however, to culturing microbes is that once the microbe is in culture, it can be experimentally used to determine its ability to grow on particular substrates. For example, while it is possible to infer a microbe may be able to grow on a particular compound from its genetic sequence, it is more scientifically compelling to directly demonstrate its growth on that compound.

Part III Outcomes

6 Methane emission estimation of the CSG holding ponds

The aims of this study were to examine the literature for evidence of emissions from gas-industry holding ponds, look to non-industry water bodies to understand the potential of waterbodies to contribute to emissions and use these data as a proxy for emissions by gas-industry water bodies, if information was found to be limited. In addition, the study sought to find key knowledge gaps, to identify potential mitigation strategies and to develop options for future work.

6.1.1 The literature on methane emissions from CSG holding ponds

Very limited data – and what exists (below) is contradictory.

- A methane emission of 221kg/h (~8840 mg/m^2/d) from a raw water pond at a CSG site in south-east Surat Basin.
- Methane emissions from a CSG holding pond in NSW were reported to be 150 mg/m^2/d in winter and 260 mg/m^2/d in summer.

6.1.2 Key results from natural/anthropogenically constructed waterbodies as sources of information about emissions

In the absence of CSG holding pond data, natural and anthropogenically constructed waterbodies would seem to be reasonable analogues to investigate the potential of the methane emissions from the CSG holding ponds in Queensland, to likely provide minimum values as starting points.

The review from of data from such waterbodies indicates the following key points:

1. Climate and/or temperature strongly affects methane emissions, through enhancing biological activity and its propensity to drive more methane to form bubbles and escape ponds as ebullitive emissions.
2. Small, shallow waterbodies are disproportionately large sources of methane emissions.
3. Nutrient load in ponds, particularly phosphorus but also dissolved organic carbon, and possibly nitrogen, are important contributors to methane emissions.

6.1.3 Applying natural pond ideas to CSG holding ponds to understand emissions potential

It is clear that pond size, temperature, organic carbon and nutrient loading impact emissions. In terms of climate most CSG water holding ponds in the Bowen and Surat basins are located in what might be described as subtropical and temperate climatic regions according to the map of climate zones for Australia from Bureau of Meteorology (http://www.bom.gov.au/climate/how/newproducts/images/zones.shtml).

In the case of size, it is important to compare the CSG holding ponds (average of 5 ha) to similarly sized natural and anthropogenically built waterbodies, therefore areas of waterbodies larger than 100 ha are not considered in the upcoming comparisons.

Simple assumption of scenarios based on small pond sizes and climate

Depending on the climate, it is therefore possible to develop some simple scenarios based on the data in Table 13, however, there are caveats.

Scenario 1: Assumes CSG holding ponds (produced water and brine) in Queensland are analogues of subtropical waterbodies. Estimated methane emissions calculated from CSG holding ponds in Queensland is around 30 mg/m^2/d.

Scenario 2: Assumes CSG holding ponds (produced water and brine) in Queensland are analogues of temperate waterbodies. Estimated methane emissions calculated from CSG holding ponds in Queensland is around 15 mg/m^2/d.

Scenario 3: Assumes half of the CSG holding ponds in Queensland are analogues of subtropical ponds, while the other 50% are temperate ponds. Estimated methane emissions is around 22 mg/m^2/d.

Table 13 Data used to calculate the estimated emissions in the scenarios. Median emission values are from Tables 2 and 3. Total CSG holding pond areas are calculated based on the Apx Table C.4.

Area category (ha)	Total CSG holding pond area (ha)	Median emissions in temperate regions (mg/m^2/d)		Median emissions in subtropical regions (mg/m^2/d)	
		Ebullitive	Diffusive	Ebullitive	Diffusive
<0.1	0.15	19	12	274	n/a
0.1-1	4.76	83	18	71	15
1-10	361	27	6.4	57	33
10-100	893	3.7	2.9	n/a	1.4

Why the scenarios are overly conservative, and identification of key knowledge gaps

One of the key assumptions for above scenarios is that natural and anthropogenically constructed waterbodies, like urban ponds, lakes or reservoirs are analogous to CSG water holding bodies. However, there are a number of reasons this unlikely. Note that points 1-3 focus only on ponds containing produced water, see point 4 for implications for brines.

1) Phosphorus. The produced water in CSG holding ponds in Queensland have elevated phosphorus levels compared to natural water bodies (0.044 mg/L). For instance, the Bowen produced water has a median phosphorus concentration of 0.3 mg/L, which is 10x that of natural ponds, and Surat produced water has phosphorus levels 2.5x higher than natural levels.
2) Organic carbon. As discussed in section 5.4.2. the reported concentrations for organic carbon in CSG holding ponds are likely to be very significant underestimates. This is because most of the carbon in the pond is in the form of ultra-fine coal particles that settles to the bottom of the pond and are not included in water chemistry testing. Not only is this carbon pool poorly understood, but it can be reasonably assumed to have a very large surface area and thus may

be more readily mobilised by microbes. Just taking the reported data, the Bowen- and Surat-derived waters have an average of 35 mg/L and 15 mg/L of total organic carbon, compared to 9mg/L in natural ponds.

3) Inorganic carbon. CSG holding ponds have very significant pools of bicarbonate (median values of 1300-1400 mg/L in produced water ponds and 6770-8295 mg/L in brine ponds). This bicarbonate can be used by chemo- or photoautotrophs (organisms that use chemical or light energy to fix carbon) as a source of carbon, thus moving some of this carbon pool into 'organic carbon' and rendering it available to be released as methane.

4) Brine pools are likely to significantly exacerbate the situations described in above in (1) and (3). For example, median phosphorus concentrations in brine ponds were 4.55 mg/L and 0.82 mg/L for the Bowen and Surat brines and are ~100x and ~20X greater than the concentrations in natural ponds. Further, brine ponds contain measured organic carbon pools that are ~4X natural levels. It should be noted that the situation for ultra-fine coal particles may not occur in brine ponds, as the fines have been removed before the desalination process (Figure 3). It should also be noted that despite their high salt content, it should not be assumed that brines do not contain halophilic (salt-loving) methanogens, as numerous halophilic methanogens are known to exist.

5) The pond area used in the calculation is significantly less than the actual pond area because there are 22 holding ponds not included in the estimates due to lack of pond area or depth information (Apx Table C.4).

6) The definition of "temperate" and "subtropical" as the literature describes it varies. For example, most climatic maps of Australia describe Sydney as being in a "temperate zone" and similarly many cities in the central USA are also described as having "temperate climates". In practice, however, the temperatures are notably colder in the central USA than in coastal Sydney and yet many lay interpretations of temperate apply and are reported in the literature.

7) The few CSG holding ponds that have been measured as mentioned in section 6.1.1 demonstrate consistently higher, but highly varied methane emissions.

To put the estimated emissions in scenarios into context, methane flux measured from an urban sewage treatment plant inlet was 28900 mg/m^2/d; an 80 ha landfill site emitted 35000 mg/m^2/d methane in the main CH_4 producing area (Day et al., 2016b) and methane emissions from forest soil being 0.31 mg/m^2/d (Feng et al., 2023) (Figure 1).

In conclusion, the true methane emissions of CSG holding ponds in Queensland is unknown. The natural waterbody estimates are likely to represent significant underestimates, despite this the concentrations proposed would still represent a significant source of methane. Further, the true emissions potential is likely higher, based on role of pond size, presence of additional carbon and nutrient inputs. It is therefore prudent to conduct further investigation by accurately quantifying methane emissions (and potentially other greenhouse gases) from CSG holding ponds in Queensland. Key knowledge gaps around organic carbon inputs to holding ponds (particularly ultra-fine coal particles) and the microbial dynamics with respect to methane production and consumption in these ponds should be addressed.

7 Future data collection activities

Elaborating on the gaps in knowledge identified in the previous section, a range of approaches could be deployed to reduce the uncertainty in emissions contributions from CSG holding ponds. In terms of future field work, wisely selecting the targeted ponds and using practical, appropriate sampling methods are critical for future methane emissions (or other greenhouse gases) investigation and closing the knowledge gaps.

7.1 CSG holding pond selection

Regarding CSG holding pond selection for future field work campaigns, several factors are recommended to be considered.

- Methane emissions from natural and anthropogenically built waterbodies are related to climate, with more methane emissions from subtropical and tropical regions than temperate regions. In Queensland, some CSG development areas are in subtropical regions, while some are in temperate regions. It is recommended to cover both climate regions for pond selection as well as obtaining more detailed seasonal variation for the sampling, say in winter and summer seasons in both climate regions.
- Depth is negatively related to methane emission from natural and anthropogenically constructed waterbodies. The calculated CSG holding pond depths range from 1.6 to 7.7 m with most ponds around or less than 5 m deep. It would be necessary to select both shallower and deeper CSG holding ponds in the future.
- Nutrients are critical inputs to methane emissions. If sampling could include ponds with different nutrient concentrations, in particular carbon and phosphorous contents, it would provide a more accurate predictor for quantifying methane emissions from the CSG holding ponds.
- Data from natural and anthropogenically constructed waterbodies in the literature are mostly freshwater bodies, while CSG holding ponds are different from fresher water ecosystems especially with respect to huge inorganic carbon content. In addition, little is known about the carbon cycle of the downstream brine ponds. It is recommended to sample from both types of holding ponds.
- CSG pond water chemistry results show that produced waters and brine in the Surat and Bowen basins are different, hence CSG holding ponds should be selected from both the Surat and Bowen basins.
- Little is known about organic carbon in the sediments within the CSG holding ponds which could be a significant carbon pool for microbes.

To limit uncertainties and avoid limitations on emission estimates from CSG holding ponds in Queensland, the selection of CSG holding ponds should be representative, rigorous, and practical. The selection should cover ponds in different basins, ponds in different climates and ponds with different purposes, different sizes (e.g., area and depth) and different nutrient contents. Because

produced water ponds and brine ponds are the two main types of CSG holding ponds, these two types of ponds would be the main target for any future study. According to the produced water treatment process (Figure 3), produced water from water management ponds (or water feed dam) represent the first step of the storage process, so water management ponds would be good candidates for sampling produced water and related methane emission (and/or other greenhouse gases). Brine ponds are good candidates for sampling brine and related methane emissions (and/or other greenhouse gases). Nutrients are a challenge factor to be included in a sampling plan.

A selection scheme is proposed in Figure 44 for further methane emission monitoring study for each basin. Basically, for a basin, we choose types of ponds (brine and produced water ponds) in each climate region; then for each type of pond, we select a smaller, shallower pond and a bigger, deeper pond. In this case, we can monitor the methane emissions related to pond depth and pond surface are which can be used to extrapolate the total methane (or other greenhouse gases) emissions from all the CSG holding ponds.

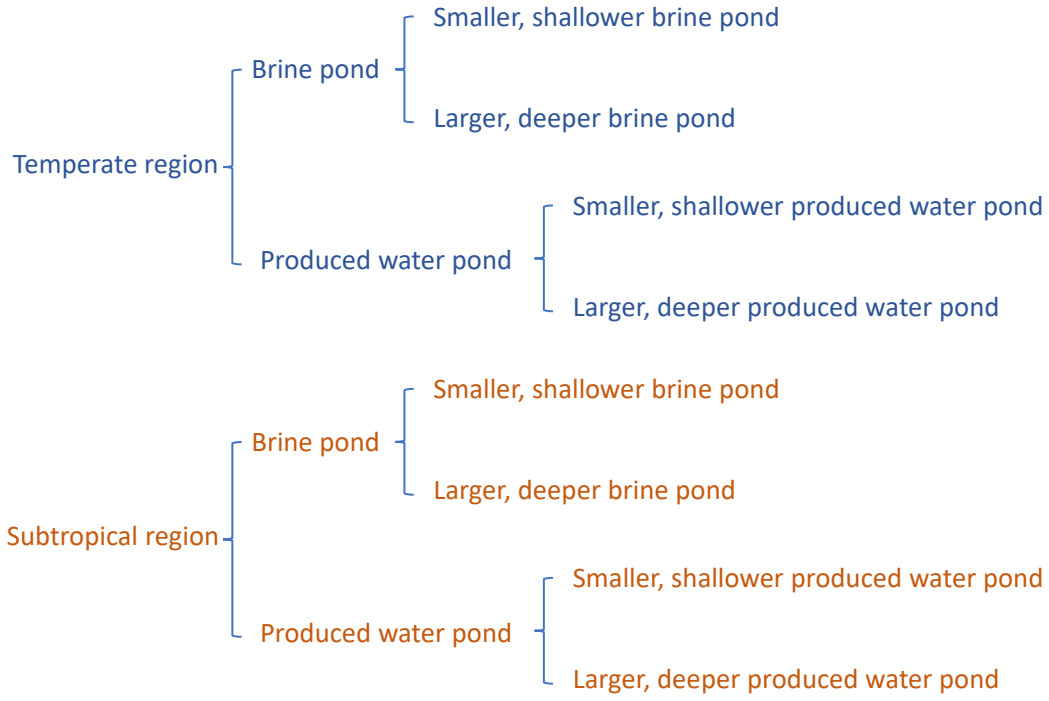


Figure 44 A proposed pond selection scheme

For other basins in Queensland, it is worthwhile to investigate the water holding pond details from the conventional oil and gas production in the Cooper Basin, CSG holding pond details in the Galilee Basin, presence of any water holding ponds in the Adavale Basin. A comprehensive sampling program will require assistance from industry to gain access and relevant background data.

7.2 Sampling

Regarding the sampling for field campaign design, several key points should be highlighted based on the literature review.

- As previously mentioned, methane emissions from natural and anthropogenically built waterbodies are primarily related to temperature, with more methane emissions from summer seasons. Seasonal sampling should be considered for future work.
- Depth is negatively related to the methane emission from natural waterbodies; hence surface and depth sampling is suggested for future work.
- Ebullition and diffusion are the dominant pathways of methane emissions from natural and anthropogenically built waterbodies, however, the relative proportion of ebullitive versus diffusive methane varies widely in waterbodies, with ebullitive: diffusive methane ratios ranging from 0.2 to 20. Capturing both ebullitive and diffusive methane emissions is critical for accurately investigating greenhouse gas emissions from the CSG holding ponds in Queensland.
- Funnel traps, floating chambers and headspace gas for dissolved gas measurement are the commonly used, simple and readily deployable methods for measuring different forms of methane emissions. However, it is noteworthy that ebullitive emissions may be localised to a specific area of the water body and also vary over day/night cycles, potentially leading to an underestimate using floating chambers or funnel bubble traps, particularly if the pond is relatively large and only a small area is sampled.
- Eddy covariance can provide estimates of emission flux over a wider area, and capture variation over long timescales, but the sensors have poor performance during wet weather. This technique is also equipment and data-processing intensive and require a fixed installation at each holding pond studied, over extended periods. This may not be feasible for multiple ponds, particularly if multiple sensors are used to obtain an ensemble mean.
- Mobile instrumentation, such as CRDS or OA-ICOS based sensors, may be a portable solution for making measurements in the field over shorter time scales, but unless they are integrated into eddy covariance methods, the usefulness of their results may be limited by shorter periods on site and limited coverage of the surface of the water body.
- ASVs and UAVs, in conjunction with newer, lightweight instrumentation may provide an opportunity to identify methane hotspots in smaller ponds, however, require specialised equipment and may not be suitable for larger ponds or over long-time scales.
- High resolution satellite sensing instrument enabled an effective spatial resolution of 50 x 50 m^2 (0.25 ha) over a targeted 12 x 12 km^2 (14400 ha) area, however, the detection limit for methane emissions may be a constraint and data processing to retrieve methane columns from satellite shortwave infrared spectra is intensive as well.

This literature review showed the limitations of emissions data availability from different published papers such as only ebullitive or diffusive emissions reported, or only total emissions reported, which is limited by the sampling methods used in the study. Each method for estimating methane flux in a water body has constraints in terms of equipment and data processing intensity, and ease of deployment. Ideally, multiple complementary methods would be used to account for ponds of different sizes and remove potential for sampling to miss localised methane hotspots. However, it may not be feasible to carry out measurements over multiple ponds if the equipment is not portable, requires lengthy fixed installation or is highly intensive to deploy.

Hence it is challenging to have a comprehensive sampling plan for investigating emissions from CSG holding ponds while trying to eliminate these limitations. In addition, temporal and spatial sampling should be considered to cover the seasonal variation and avoid location bias.

7.2.1 Methane emissions sampling

Ebullitive methane

This literature review showed that both floating chamber and funnel bubble trap can be used for sampling ebullitive methane emissions, however, the measurement from floating chambers requires extra calculations to determine the proportions of ebullitive versus diffusive emissions. A few options could be explored for future study.

- Floating chamber with automated gas sampling: This will involve a change to chamber design, electronics and sensors assembling as well as programming to log the real-time data. A design from Thanh Duc et al. (2020) could capture both ebullitive and diffusive emissions and declared to be an inexpensive device. This method can monitor over long periods of time. Some other automated systems may still require manual maintenance and gas release to maintain pressure (Varadharajan et al., 2010).
- Floating chambers with manual gas sampling: This option will require the design of a floating chamber and manual gas sampling. A syringe withdrawing gas from the floating chamber requires easy access to a laboratory close by so that the gas can be analysed in a timely manner. However, if the gas can be sampled into isotubes, it can then be stored for a few months safely without any loss.
- Funnel bubble trap with manual gas sampling: This method is comparatively cost-effective, and easy to perform for the greenhouse estimation from the ponds. There is an in-house bubble trap in our lab which was set up for a previous CSIRO study (Day et al., 2016b)., 2016) which could be used for this study. More bubble traps could be made with this requirement. In this bubble trap design, the gas can be sampled into isotubes in which gas can be safely stored for months; hence it will not be limited by laboratory accessibility. It is suggested the sampling duration using a funnel bubble trap should be less than 20 h to avoid diffusion being captured (Malyan et al., 2022).

It is recommended that the funnel bubble trap method could be used for sampling ebullitive methane from the CSG holding ponds since it is readily available and easy to build more with the same requirement. The number of bubble traps needed to set up for a particular pond is subject to the pond size and the accessibility to the pond. Since the ebullition happens quite sporadically in a pond, sampling only one location in a pond may underestimate the emissions.

Diffusive methane

Headspace gas from water samples was often used to analyse diffusive methane emissions. The literature review shows that using isoflasks to sample water for dissolved methane analysis turned out to be more accurate than by conventional vial method (Pearce et al., 2023) which also can be used for isotope analysis of methane. The water sample with its headspace gas can be stored for two months without compositional and isotopic values being altered.

Atmospheric methane

This work has determined that the current known air sampling methods for methane emissions analysis are either more equipment-heavy or data-processing intensive, requiring multiple physical and meteorological measurements in addition to gas monitoring. The Eddy covariance (EC) technique is increasingly used for long-term monitoring of terrestrial and lake-dominated landscapes, but it is expensive in terms of equipment (Deemer et al., 2016; Vesala et al., 2014) and will require permission to set up such a thing next to a CSG holding pond. Satellite sensing method could be explored since capability is available in CSIRO even though connecting top-down information on methane emissions to the improvement of bottom-up emissions inventory remains a challenge.

Additionally, another in-house method has been designed to analyse not only concentration of methane but also carbon and hydrogen isotopes of methane in collected air. Pre-vacuumed air canisters can be used for sampling the air at the CSG holding pond sites, and an in-house developed concentrator connected with natural gas analyser or gas chromatography-isotope ratio-mass spectrometer can be used to analyse compositional and isotopic composition of the air samples.

7.2.2 Water and sediment sampling

As previously mentioned (Section 5.4.2), a potentially important carbon pool is from the materials settled at the bottom (sediment) of the ponds, and these have not been characterised previously, hence future work should cover both water and sediment sampling to close the knowledge gap.

Water and sediment analysis for the greenhouse emission study can provide not only water chemistry or sediment chemistry information but also, more importantly, microbial community data in the CSG holding ponds. The presence of methanogens and methanotrophs plays a critical role on methane generation and mitigation.

For a CSG holding pond, it is speculated that the water chemistry and microbial community might be different between the surface and bottom of the pond and between inlet and drainage point. Proposed sampling locations are illustrated in Figure 45. The top view picture showing the pond surface has two notional sampling locations with one close to the inlet and one at the outlet. The side view picture shows a vertical profile of the surface and water depth sampling (red dots) and sediment sampling points at the bottom (grey dots).

Sampling for microbial analysis requires triplicate sampling at least at the designated locations. Live samples might be required for culturing the microbes for methane mitigation purpose. Live water sampling is more involved because the samples need to be kept anoxic for the bottom samples.

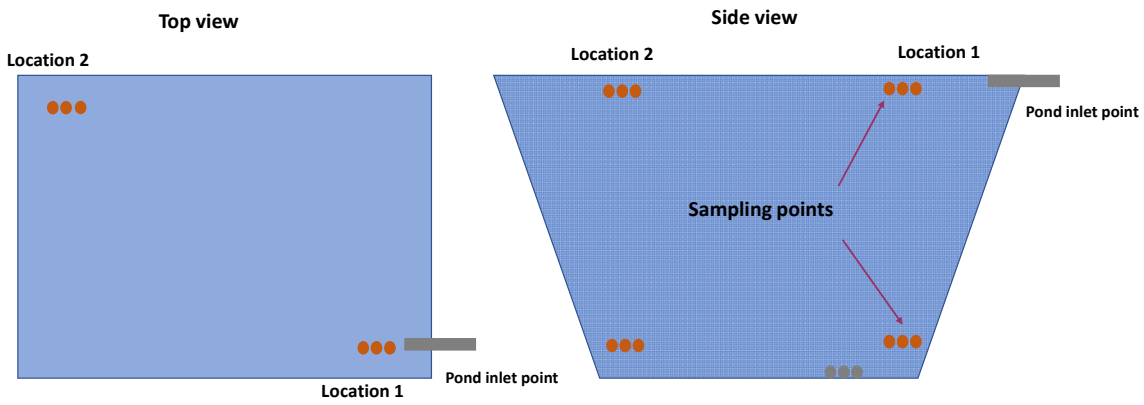


Figure 45 Proposed sampling locations in a CSG holding pond

While the above provide a range of approaches to fill the data and knowledge gaps for emissions from CSG holding ponds, it is acknowledged that there may need to be the need to take two different approaches to the actual field campaigns for sampling.

Case 1 – focus on methane emissions only from a larger number of ponds to increase the number of data points from <5.

Case 2 – focus on an intense study of one or two ponds to incorporate, water chemistry, microbial activity and related controls on methane generation/consumption.

Each of the above provide unique and new information on the potential for methane emissions from CSG ponds that is essential to be able to identify the contribution of CSG holding ponds in Queensland that could be reduced as part of the Global Methane Pledge for 2030.

8 Potential methane emission mitigation

In spite of the limited amounts of data available on methane emissions in CSG holding ponds, it is possible to draw on parallel experience and capability to identify potential mitigation strategies that could be deployed to reduce overall emissions in the future.

8.1 Biological approaches

Of the methane that is produced globally, a very significant portion is already mitigated through the process of methanotrophy. In lakes, for example, it has been demonstrated that methanotrophs growing in the oxic depths of the pond, rapidly respond to increasing methane levels through growth and replication and consume most of the available methane (Mayr et al., 2020). This consumption represents a significant portion of the produced methane. For example, studies have demonstrated that between 46-98% of methane produced in water bodies can be oxidised by methanotrophic microbes (Encinas Fernández et al., 2014; Kankaala et al., 2013; Mayr et al., 2020; Schubert et al., 2012). The ability of biological approaches to scale up and scale down as methane fluxes change is a clear advantage of biological approaches over chemical applications.

Biological emission mitigation efforts should focus on factors that enhance rates of methanotrophy. There are a number of complementary and interconnected considerations that are worth further exploration:

1) Dissolved oxygen in the water

The majority of, but not all, methantrophs are aerobic. Aerobic methantrophs require oxygen to grow. Methanotrophs are ubiquitous in virtually all environments. When methane upwells from anaerobic activity (or from geological processes) this upwelling methane meets oxygen that is diffusing down into the environment (whether it be in water columns, soils, sediments or other situations inside living organisms). Increasing the oxygen concentration in the water column thus has two advantages, firstly it would increase the inhabitable zones for aerobic methanotrophs and secondly, it would limit the regions of the ponds that are conducive to methanogenesis (i.e., anoxic zones). Mechanisms and options for aerations are discussed below, but it is noteworthy that such mechanisms may also act to accelerate methane emissions (through increasing the velocity of escaping methane).

2) Pond depth

Pond depth is negatively associated with emissions. This is likely largely a result of the longer residency time of the methane in deeper ponds. Put simply, there is more opportunity in a deeper pond for methane to be consumed by methanotrophs. Like dissolved oxygen, however, deeper ponds result in more anoxic zones, which in turn, may be reservoirs for higher levels of methanogenesis. Regardless, this overall positive trend in deeper ponds, likely outweighs the 'methane-producing' costs of such ponds. Currently, the average pond depth for CSG holding water ponds is ~4-5 m (see data in Section 5.3) and it may be that engineering ponds to increase the depth of the water may assist in controlling methane emissions.

3) Macronutrient and electron-acceptor considerations

Microbial activity in ponds is largely constrained by availability of important macronutrients, mostly (but not exclusively) the availability of nitrogen and phosphorus. The former is more difficult to control than the latter as nitrogen is abundant in atmospheric air and, in the absence of dissolved, easily accessible forms of nitrogen, microbial communities will often use nitrogen fixation as a primary means of obtaining this nutrient. Regardless, limiting access to these two nutrients (or even just phosphorus) severely constrains which microbial processes occur. This is because phosphorus is key to making not only phospholipids for new cell membranes, but also for critical cellular molecules like DNA or RNA. It may be possible to remove these macronutrients through biologically scrubbing or chemical processes that target and remove these chemicals.

Changes in available electron acceptors may also affect the rates of methanogenesis from ponds. This concept likely requires a small introduction. In essence, living organisms undertake a process called respiration. In humans (and all other animals) this takes the form of oxidising glucose (or other energy carrying molecules) and reducing oxygen. In this process, oxygen is called the electron acceptor, and glucose (the thing that is oxidised) is the electron donor. Bacteria and archaea also respire, but they can use electron acceptors other than oxygen. This includes compounds such as nitrate, iron, manganese, sulfate, sulfur and carbon dioxide. There is also different amount of energy yielded from reducing different electron acceptors. The presence of an electron acceptor that yields more energy, often shifts the community away from the use of lower energy-yielding electron acceptors (mostly through improved competitiveness). For example, in oil reservoirs, nitrate is sometimes added to inhibit souring (i.e., the reduction of sulfur or sulfate). This works because the anaerobic reduction of nitrate (to N_2O and subsequently to N_2) yields significantly more energy than the reduction of sulfur or sulfate to H_2S and results in organisms that can use nitrate being energetically favoured in these environments. It thus may be possible to add other electron acceptors to CSG water ponds that may alter emissions potential of these ponds. Some of these approaches may create other issues. For example, the addition of nitrate may enhance rates of eutrophication in these ponds and exacerbate activity associated with having a macronutrient (e.g., nitrogen) being suddenly available at higher concentrations.

4) Methanogenesis inhibitors

Finally, it may be possible to add chemicals to ponds to directly inhibit methanogenesis in these environments. These compounds are already used in agriculture (as ruminant feed supplements) and may be of assistance in reducing biological methanogenesis in CSG (or other energy related) holding ponds.

8.2 Engineering approaches

Currently, geological methane is separated from the water phase inside the gas separator. However, observations downstream of this separator indicate that some geological methane (saturated in the produced water) has yet to evolve from the water phase. It may be that through improving the engineering in the separator or downstream equipment it may be possible to remove the small amount of methane that occurs post-separation. Options may include desorbing the gas under a gentle vacuum or the use of a spray tower and packed column filter to remove the last of the dissolved methane prior to that material entering the holding pond. Such approaches

likely yield a gas stream with relatively small concentrations of methane. Uses or disposal of this methane would, in turn, require other engineering solutions.

It would seem likely, however, that most of the methane is already removed in the separator (as discussed earlier) but still requires quantification for reporting.

Other engineering solutions may include additional filters to remove ultra-fine coal particulates or use of materials in the construction of the pond that assist in limiting methanogenesis.

Appendix A Background to selected geological basins of Queensland

Queensland plays host to a range of different sedimentary basins, including the Surat Basin, Bowen Basin, Galilee Basin, Adavale Basin, and the Cooper Basin. These basins have come under considerable attention for their capacity to either produce or have the potential to produce conventional and/or unconventional oil and gas resources. Below, we briefly outline their geological settings and provide a background to their origin and characteristic features.

The Bowen Basin (Figure A.1) covers an area of approximate 160,000 km^2 (Cadman et al., 1998) and forms part of the larger Permo-Triassic Sydney-Gunnedah-Bowen Basin system (Scheibner, 1999). The Bowen Basin formed during the Late Carboniferous and Early Permian and ceased during the Late Triassic with a 30 Ma-year period of erosion marking the divide between Bowen and Surat basins (Cadman et al., 1998). Sedimentary rocks within the Bowen Basin are predominantly sandstones, shales, claystones and coal intermingled with extrusive volcanics and the intrusion of plutons (Danis et al., 2012). Coal seams in the Bowen Basin are classified into four distinct groups with six source rock units recognized to account for the bulk of the generated hydrocarbons: Moolayember Formation, Baralaba Coal Measures, Burunga Formation, Banana Formation, Flat Top to Buffel formations and the Reids Dome beds (Boreham et al., 1996; Carmichael and Boreham, 1997; Mallett et al., 1995). These coal seams are thick and laterally continuous. Exploration for coal seam gas in the Permian coals of the Bowen Basin started in late 1970s (Troup et al., 2018) with the first commercial production began in the Dawson River CSG area near Moura in 1996 (DNRM, 2017). There are two main regions of CSG exploration and development in the Bowen Basin with higher permeability targets in the southern-central Bowen Basin and lower permeability targets in the north (Troup et al., 2018). Up until 2014-2015, the Bowen Basin had been the largest cumulative CSG producing basin with the Permian coal measures being the most productive units, forming more than 90% of the oil and around 65% of the thermogenic gas (Shaw et al., 2000; Towler et al., 2016).

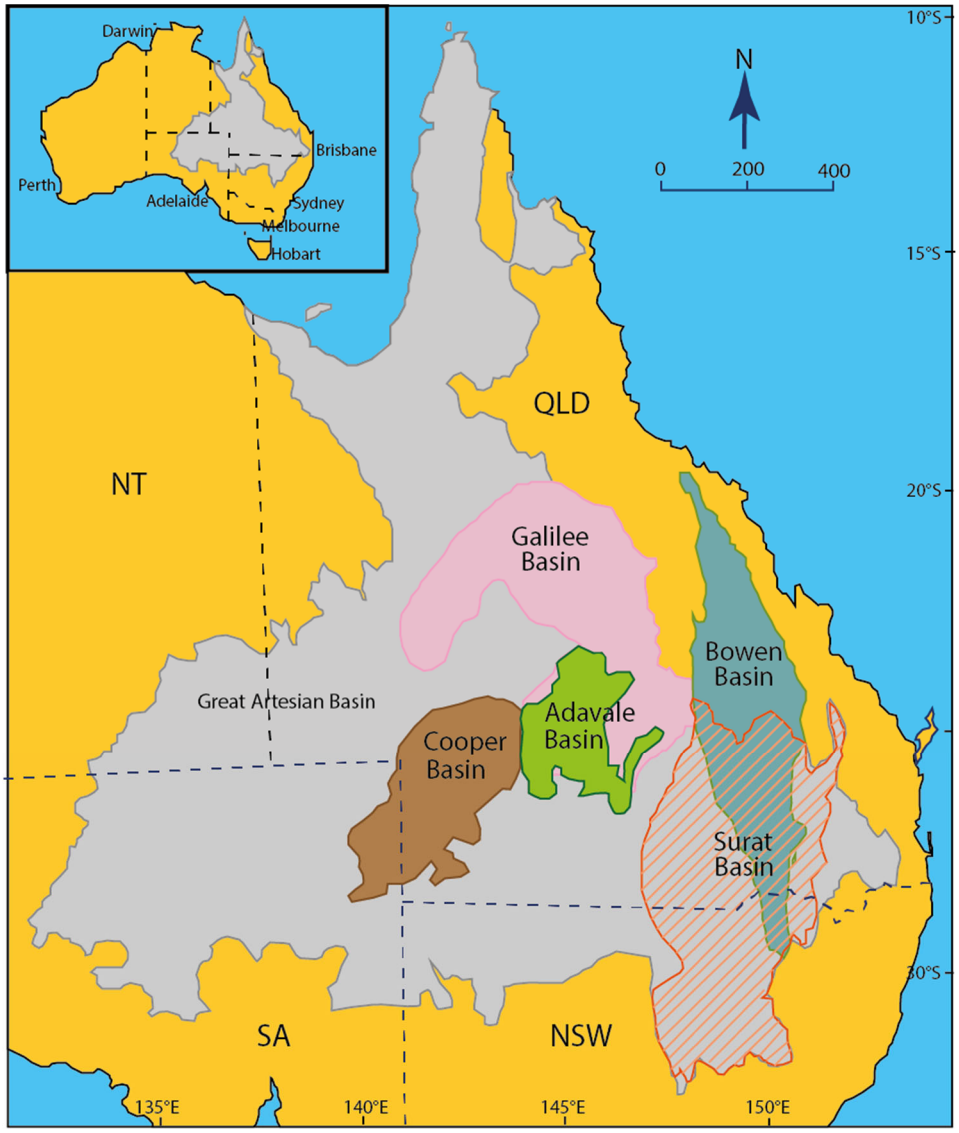
The Triassic-Cretaceous Surat Basin (Figure A.1) overlies the southern part of the Bowen Basin and formed through further tectonic activity (Korsch and Totterdell, 2009) and ceased at about 95 Ma (Yago, 1996). It is a large intra-cratonic basin with up to 2.5 km sedimentary infill (a sedimentary process of filling a basin). The Middle to Upper Jurassic Walloon Coal Measures contain the most significant coal resources in the Surat Basin. The coal seams are thin (1-30 cm) with limited lateral continuity (Morris and Martin, 2017). In the early 2000s, the Jurassic Walloon Coal Measures in the Surat Basin began to the targeted by CSG exploration. From 2011-2012 on, CSG production from the Surat Basin has been higher than that from the Bowen Basin and in 2015-2016, production from the Surat Basin was more than four times that of the Bowen Basin (DNRM, 2017). The coal seam gas in the Walloon Coal Measures is mainly biogenic in origin (Faiz and Hendry, 2006), containing predominantly methane (>98%; air-free basis) with minor CO_2 and N_2 (Hamilton et al., 2012).

The Permian to Triassic Cooper Basin (Figure A.1) spans the borders of both Southwestern Queensland and South Australia. With an infill of 2.5 km of sediments, the Cooper Basin contains deposits of non-marine depositional sequences, including coal (Jensen-Schmidt et al., 2006). This basin encompasses an area of approximately 130,000 km^2 and has been the premier onshore

hydrocarbon gas producing basin in Australia, which supplies the East Coast gas market (Gravestock et al., 1998). Most of the activity hitherto conducted in this field concerned conventional petroleum exploration in South Australia. Unconventional resources have also been confirmed and include shale gas, tight gas and CSG. However, CSG-related activity in this basin have thus far been restricted to the South Australia side and no such activity in Queensland is currently being undertaken (Salmachi et al., 2021).

The crescent-shaped Galilee Basin (Figure A.1), situated in the western portion of Queensland, covers an area of approximately 247,000 km^2 (Scott et al., 1995). Numerous tectonic theories have been proposed for the origin of this basin (I' Anson, 2013; Jackson et al., 1981; Waschbusch et al., 2009), with sedimentation being initiated from the Late Carboniferous and terminating by the Middle Triassic. Sedimentary successions that comprise the infill of this basin are composed of sandstone, mudstone and coal from a range of terrestrial and marginal marine palaeoenvironments. The Galilee Basin contain extensive resources of Permian black coal. The Late Permian Betts Creek Beds are the principal targets for the current CSG exploration program (GalileeEnergy, 2022).

The Early Devonian to Early Carboniferous Adavale Basin (Figure A.1) is situated in central Queensland with an area occupying approximately 60,000 km^2 and largely underlain by the Galilee Basin (Boreham and Boer, 1998; Troup and Talebi, 2019). After basin formation through tectonic activity, volcanic, terrestrial, and marine-derived sedimentary successions were deposited. While the Adavale Basin has a confirmed petroleum system producing wet gas, it is currently an under-explored, frontier petroleum basin and subject to renewed exploration interest (Boreham and Boer, 1998; Troup and Talebi, 2019).



Apx Figure A.1 Locations of the main geological basins in Queensland

Appendix B Data analysis methods

B.1 Water chemistry and emissions data from natural and anthropogenically constructed waterbodies

The collected emission data from natural and anthropogenically built waterbodies are separated into three groups, ebullitive methane emissions, diffusive methane emissions and combined (ebullitive and diffusive combined) methane emissions in Apx Table C.1, Apx Table C.2 and Apx Table C.3. It is noted that a unit, mg/m^2/d, is used in this study to unify the different emissions units reported in different papers for easy comparison.

To summarize each type of emissions, seven waterbody area ranges are divided as <0.1 ha, 0.1-1 ha, 1-10 ha, 10-100 ha, 100-1000 ha, 1000-10000 ha, >10000 ha (Tables 3, 4 and 5). In each area range, climates of waterbodies located are separated. Depending on the data availability, there could be one climate type or more climate types in each area range. In each climate zone, minimum, maximum and median values of the methane emissions were calculated. Boxplots were drawn to show the distribution of different forms of methane emissions in different climates in each area category. It is noteworthy that emissions without related area information are not included in this grouping for the tables. But the boxplots included all the emissions data listed in the tables in the Appendix C.

B.2 Plotting and analyses

Statistical analyses

In section 4.3, T test and ANOVA test are used to compare two groups of data and three or more groups of data.

In section 4.5, correlations and associated p-values were calculated using Pearsons R in SciPy v. 1.7.3.

Scatter plots

In section 4.5, scatter plots were all drawn using Matplotlib v. 3.5.1 and Seaborn v. 0.11.2 for Python v. 3.7.4.

Violin plots

Violin plots combine the characteristics of a box plot with a density chart. The integral shape of the violin chart with the box represents the estimated density of the data distribution of a given physicochemical phenomena in a sample. White dots in the violin plots are the median values of the dataset.

Violin plots were drawn using Matplotlib v. 3.5.1 and Seaborn v. 0.11.2 for Python v. 3.7.4. To remove outliers, for each chemical z-scores were calculated from the data using a custom Python script (Scipy v. 1.7.3 and numpy v. 1.21.5, then those data points with absolute z-scores in excess of 3 were removed. The range of these excluded outliers, and the total number of outliers is recorded in the title of each plot.

Box plots

Box plots were all drawn using Matplotlib v. 3.5.1 and Seaborn v. 0.11.2 for Python v. 3.7.4. The rectangle in the plot represents the interquartile range, with the median shown as a line horizontally crossing the rectangle, max and min are shown by whiskers. Outliers, where they occur are shown as dots or asterisks above or below the max/min whiskers.

Appendix C Tables

Apx Table C.1 Ebullitive methane emissions and related water chemistry of natural and anthropogenically built waterbodies

Name	Location	Area (ha)	Temperature (°C)	pH	Depth (m)	DOC (mg/l)	TN (mg/l)	TP (ug/l)	Ebullitive emissions (mg/m2/d)	Climate	References
Lake North Gate	North America	0.3	23			21.3		14.25	2.5	Temperate	Bastviken et al., (2004)
Lake Tuesday	North America	0.9	23			7.6		8.98	3.2	Temperate	Bastviken et al., (2004)
Lake Paul	North America	1.7	23			3.1		11.77	7.3	Temperate	Bastviken et al., (2004)
Lake East Long	North America	2.3	23			12.1		37.78	7.3	Temperate	Bastviken et al., (2004)
Lake Peter	North America	2.7	23			4.5		7.74	12.2	Temperate	Bastviken et al., (2004)
Lake Ward	North America	2.7	23			6.5		26.63	27.4	Temperate	Bastviken et al., (2004)
Lake Crampton	North America	25.8	23			3.7		7.43	2.3	Temperate	Bastviken et al., (2004)
Lake Brown	North America	32.9	23			7.9		35	3.9	Temperate	Bastviken et al., (2004)
Lake Roach	North America	45	23			2.7		5.88	0.3	Temperate	Bastviken et al., (2004)
Lake Hummingbird	North America	0.8	23			22		33.76	3.2	Temperate	Bastviken et al., (2004)
Lake Morris	North America	5.9	23			12.7		19.2	44.6	Temperate	Bastviken et al., (2004)
Pond D	Stordalen Mire, Sweden	0.0013	10	4.5	0.41				3.6	Temperate	Burke et al., (2019)
Pond C	Stordalen Mire, Sweden	0.0024	10.2	4.3	0.35				4.5	Temperate	Burke et al., (2019)
Pond E	Stordalen Mire, Sweden	0.0083	5.8	3.8	0.85				53.4	Temperate	Burke et al., (2019)
Pond F	Stordalen Mire, Sweden	0.0135	8	4.3	0.43				40.9	Temperate	Burke et al., (2019)
Pond H	Stordalen Mire, Sweden	0.0161	12.2	5.9	0.41				26.5	Temperate	Burke et al., (2019)
Pond G	Stordalen Mire, Sweden	0.0451	11.6		0.47				11.6	Temperate	Burke et al., (2019)

Priest Pot	Eurasia	1				599.3	182	Temperate	Casper et al., (2000)
Lake Ontario	North America	1896000		12		8.05	1.5	Temperate	Chau et al., (1977)
Curua-Uná	Brazil	7200	6				65	Tropical	Duchemin et al., (2000)
Curua-Uná	Brazil	7200	6				12	Tropical	Duchemin et al., (2000)
Pinjarra 3	Queensland, Australia	0.021					143	Subtropical	Grinham et al., (2018)
Indooroopilly	Queensland, Australia	0.0436					274	Subtropical	Grinham et al., (2018)
Gatton 6	Queensland, Australia	0.0446					724	Subtropical	Grinham et al., (2018)
Greenbank	Queensland, Australia	0.0575					166	Subtropical	Grinham et al., (2018)
Mt Cootha	Queensland, Australia	0.058					1405	Subtropical	Grinham et al., (2018)
Gatton 3	Queensland, Australia	0.104					905	Subtropical	Grinham et al., (2018)
Mt Larcom 2	Queensland, Australia	0.1256					45	Subtropical	Grinham et al., (2018)
St Lucia 3	Queensland, Australia	0.1755					49	Subtropical	Grinham et al., (2018)
Gatton 4	Queensland, Australia	0.189					55	Subtropical	Grinham et al., (2018)
Pinjarra 2	Queensland, Australia	0.1943					59	Subtropical	Grinham et al., (2018)
Gatton 2	Queensland, Australia	0.345					170	Subtropical	Grinham et al., (2018)
St Lucia 2	Queensland, Australia	0.429					83	Subtropical	Grinham et al., (2018)
Mt Larcom 1	Queensland, Australia	0.5025					37	Subtropical	Grinham et al., (2018)
Fig Tree park	Queensland, Australia	0.8357					301	Subtropical	Grinham et al., (2018)
Mt Larcom 3	Queensland, Australia	1.6093					17	Subtropical	Grinham et al., (2018)
Lake Alford	Queensland, Australia	2.1689					29	Subtropical	Grinham et al., (2018)
St Lucia 1	Queensland, Australia	2.273					282	Subtropical	Grinham et al., (2018)
Gatton 1	Queensland, Australia	2.59					590	Subtropical	Grinham et al., (2018)
Gatton 5	Queensland, Australia	3.045					122	Subtropical	Grinham et al., (2018)
Oxenford	Queensland, Australia	3.694					94	Subtropical	Grinham et al., (2018)
Port Precinct	Queensland, Australia	3.828					57	Subtropical	Grinham et al., (2018)
Pinjarra 1	Queensland, Australia	5.678					15	Subtropical	Grinham et al., (2018)

Postilampi	Eurasia	3			58	Temperate	Huttunen et al., (2003)
Reservoir Porttipahta	Eurasia	21400			3.5	Temperate	Huttunen et al., (2003)
Reservoir Lokka	Eurasia	41700			12.35	Temperate	Huttunen et al., (2003)
Reservoir Lokka	Eurasia	41700			33.7	Temperate	Huttunen et al., (2003)
Gatun Lake	Panama	42500	30	0.5-0.7	607	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	0.6-0.7	297	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	0.5-1.0	1978	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	0.6-1.0	1908	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	0.7-0.9	3635	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	0.7-0.9	1571	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.0-1.1	309	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.2-1.3	812	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.4-1.5	730	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.4-1.5	548	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.4-1.5	621	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.4-1.5	769	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.4-1.6	1186	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.4-1.6	673	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.4-1.6	871	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.6-1.7	924	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.6-1.7	1229	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.6-1.7	1795	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.4-2.6	337	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	1.4-2.6	1352	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	3.0-3.7	262	Tropical	Keller and Stallard, (1994)

Gatun Lake	Panama	42500	30	3.0-4.4	127	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	3.7-4.1	641	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	3.9-4.2	172	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	3.9-4.3	161	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	3.9-4.5	186	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	3.9-4.5	429	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	3.9-4.5	619	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	4.2-4.8	480	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	4.2-5.0	168	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	4.2-5.0	645	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	4.2-5.0	831	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	3.9-5.4	188	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	4.6-4.9	1148	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	4.6-4.9	792	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	4.6-4.9	524	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	5.1-5.4	478	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	5.1-5.5	735	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	5.1-5.5	161	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	5.1-5.5	497	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	6.7-7.1	42	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	6.9-7.1	207	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	6.9-7.1	40	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	6.9-7.1	103	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	8.6-9.1	13	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	9.6-9.8	29	Tropical	Keller and Stallard, (1994)
Gatun Lake	Panama	42500	30	9.6-9.8	26	Tropical	Keller and Stallard, (1994)

Lake Mirror	North America	15				4			2.4	Temperate	Mattson and Likens, (1993)
Pond in Linköping	Sweden	0.12			1.2				122	Temperate	Natchimuthu et al., (2014)
Streams	Berlin, Germany	0.79							66	Temperate	Herrero Ortega et al., (2019)
Ponds	Berlin, Germany	2.11							300	Temperate	Herrero Ortega et al., (2019)
Rivers	Berlin, Germany	21.4							109	Temperate	Herrero Ortega et al., (2019)
Lakes	Berlin, Germany	29.7							100	Temperate	Herrero Ortega et al., (2019)
Ponds	Minnesota, The USA	1		7.5	1	12	0.8	147	484	Temperate	Rabaey and Cotner, (2022)
Lake Schutsloterwiede	Netherlands	141		8.4	1.2				444	Temperate	Schrier-Uijl et al., (2011)
Lake Belterwiede	Netherlands	613		8.4	1.8			12.6	62	Temperate	Schrier-Uijl et al., (2010)
Lake Nieuwkoopse plas	Netherlands	676		8.2	2.5			10.3	96	Temperate	Schrier-Uijl et al., (2010)
Lake Reeuwijkse plas	Netherlands	927		9	2.1			11.3	72	Temperate	Schrier-Uijl et al., (2010)
Lake Vinkeveense plas	Netherlands	1079		8.4	2.4			14	57	Temperate	Schrier-Uijl et al., (2010)
Lake Wintergreen	Michigan, North America	15							337	Temperate	Strayer and Tiedje, (1978)
Urban pond	Gelderland, Netherlands	0.4635	13	7.2					100	Temperate	van Bergen et al., (2019)
Lake Donghu	Wuhan, China	2790	28		2.5	7.5			23.3	Subtropical	Xing et al., (2005)

Apx Table C.2 Diffusive methane emissions and related water chemistry of natural and anthropogenically built waterbodies

Name	Location	Area (ha)	Temperature (°C)	pH	Depth (m)	DOC (mg/l)	TN (mg/l)	TP (ug/l)	Diffusive emissions (mg/m2/d)	Climate	References
Urban ponds	Silkeborg, Denmark	7.656	11.9	7.6			1.07	190	15	Temperate	Audet et al., (2020)
Lillsjön	Eurasia	2	22			19.8		15.5	0.43	Temperate	Bastviken et al., (2002)
Illersjön	Eurasia	3.9	22			9.4		38.7	9.64	Temperate	Bastviken et al., (2002)
Mårn (south sub-basin)	Eurasia	4.5	22			17.9		52	1.04	Temperate	Bastviken et al., (2002)
L1 Ladario	South America	6	30	5.9		10.88			12.99	Tropical	Bastviken et al., (2010)
N7a Nhumirim	South America	10	34	10.7		5.76			10.59	Tropical	Bastviken et al., (2010)
L4 Ladario	South America	10	30	6.6		9.73			13.95	Tropical	Bastviken et al., (2010)
L2 Ladario	South America	12	30	6		3.58			11.23	Tropical	Bastviken et al., (2010)
N6b Nhumirim	South America	12	27	8.1		14.05			7.7	Tropical	Bastviken et al., (2010)
N8a Nhumirim	South America	16	29.5	9		3.82			11.07	Tropical	Bastviken et al., (2010)
L3 Ladario	South America	35	30	6.3		1.52			11.55	Tropical	Bastviken et al., (2010)
BB National Park	South America	3630	30.5	6.2		0.74			8.02	Tropical	Bastviken et al., (2010)
TR National Park	South America	7140	31	6		0.75			10.43	Tropical	Bastviken et al., (2010)
Tereza Ladario	South America		24	6.7		1.3			2.89	Tropical	Bastviken et al., (2010)
Presa Ladario	South America		27	6.6		2.85			13.79	Tropical	Bastviken et al., (2010)
Lobo Ladario	South America		24	6.5		6.45			28.55	Tropical	Bastviken et al., (2010)
N14 Nhumirim	South America		29.5	5.7		8.03			6.26	Tropical	Bastviken et al., (2010)
N19a Nhumirim	South America		30	5.3		8.8			11.71	Tropical	Bastviken et al., (2010)
Belém, Ladario	South America		26	6.5					3.21	Tropical	Bastviken et al., (2010)
Bracinho Ladario	South America		26	6.7					29.03	Tropical	Bastviken et al., (2010)
Lake North Gate	North America	0.3	23			21.3		14.2	2.58	Temperate	Bastviken et al.(2004)
Svarttjärn	Eurasia	0.7	22			18.8		13.9	12.34	Temperate	Bastviken et al., (2004)

Lake Hummingbird	North America	0.8	23		22	33.8	2.47	Temperate	Bastviken et al., (2004)
Lake Tuesday	North America	0.9	23		7.6	9	9.78	Temperate	Bastviken et al., (2004)
Lake Paul	North America	1.7	23		3.1	11.8	10.37	Temperate	Bastviken et al., (2004)
Lake East Long	North America	2.3	23		12.1	37.8	3.92	Temperate	Bastviken et al., (2004)
Lake Peter	North America	2.7	23		4.5	7.7	6.68	Temperate	Bastviken et al., (2004)
Lake Ward	North America	2.7	23		6.5	26.6	6.18	Temperate	Bastviken et al., (2004)
Skottjärn	Eurasia	2.8	22		20.5	14.9	11.98	Temperate	Bastviken et al., (2004)
Rågåstjärn	Eurasia	4	22		5.8	8.7	7.21	Temperate	Bastviken et al., (2004)
Gransjön	Eurasia	4.5	22		11.6	22.9	3.55	Temperate	Bastviken et al., (2004)
Lake Morris	North America	5.9	23		12.7	19.2	7.21	Temperate	Bastviken et al., (2004)
Lövtjärn	Eurasia	8.5	22		6.6	9.6	3.98	Temperate	Bastviken et al., (2004)
Klintsjön	Eurasia	10	22		4.7	6.5	2.33	Temperate	Bastviken et al., (2004)
Ljustjärn	Eurasia	13	22		4.4	8.4	3.09	Temperate	Bastviken et al., (2004)
L Sångaren	Eurasia	24	22		7.4	12.1	1.15	Temperate	Bastviken et al., (2004)
Lake Crampton	North America	25.8	23		3.7	7.4	2.12	Temperate	Bastviken et al., (2004)
Gyslättasjön	Eurasia	26.3	22		13.4	20.1	2.87	Temperate	Bastviken et al., (2004)
Skärshultssjön	Eurasia	29.2	22		15.9	22	1.26	Temperate	Bastviken et al., (2004)
Lake Brown	North America	32.9	23		7.9	35	4.84	Temperate	Bastviken et al., (2004)
Bisen	Eurasia	43.3	22		8.3	8.7	0.69	Temperate	Bastviken et al., (2004)
Lake Roach	North America	45	23		2.7	5.9	1.65	Temperate	Bastviken et al., (2004)
Grunnen	Eurasia	48	22		17.2	22	3.39	Temperate	Bastviken et al., (2004)
Fiolen	Eurasia	150	22		6.8	13	0.59	Temperate	Bastviken et al., (2004)
Priest Pot	Eurasia	1				599.3	5.94	Temperate	Casper et al., (2000)
Curua-Uná	Brazil	7200		6			16	Tropical	Duchemin et al., (2000)
Curua-Uná	Brazil	7200		6			20	Tropical	Duchemin et al., (2000)
Laforge-1	Quebec, Canada	96000		3			8	Temperate	Duchemin et al., (1995)

La Grande-2	Quebec, Canada	283500			22				3	Temperate	Duchemin et al., (1995)
Lake Mendota	North America	3937				10		8.1	5.97	Temperate	Fallon et al., (1980)
Stormwater pond 14	Florida, The USA	0.622			7.2	11.2	1.2		129	Subtropical	Goeckner et al., (2022)
Stormwater pond 23	Florida, The USA	0.79			3.4	12.1	0.91		14.6	Subtropical	Goeckner et al., (2022)
Stormwater pond 15	Florida, The USA	0.885			4.7	8.1	0.97		29.3	Subtropical	Goeckner et al., (2022)
Stormwater pond 34	Florida, The USA	1.31			1.8	16.8	1.3		44.3	Subtropical	Goeckner et al., (2022)
Stormwater pond 18	Florida, The USA	1.88			5.6	9.3	1.1		21.5	Subtropical	Goeckner et al., (2022)
Stormwater ponds	Virgina, The USA	40.73	25	7.3	3				362	Temperate	Gorsky et al., (2019)
Postilampi	Eurasia	3			3.2				59.36	Temperate	Huttunen et al., (2003)
Postilamp	Eurasia	3			3.2				59.36	Temperate	Huttunen et al., (2003)
Postilampi	Eurasia	3			3.2				78.61	Temperate	Huttunen et al., (2003)
Heinälampi	Eurasia	9.8			5				5.9	Temperate	Huttunen et al., (2003)
Mäkijärvi	Eurasia	20			3.4				2.89	Temperate	Huttunen et al., (2003)
Mäkijärvi	Eurasia	20			3.4				1.77	Temperate	Huttunen et al., (2003)
Vehmasjärvi	Eurasia	41			3.9				5.6	Temperate	Huttunen et al., (2003)
Vehmasjärvi	Eurasia	41			3.9				1.76	Temperate	Huttunen et al., (2003)
Kevätön	Eurasia	407			2.3				81.8	Temperate	Huttunen et al., (2003)
Kevätön	Eurasia	407			2.3				51.3	Temperate	Huttunen et al., (2003)
Robert-Bourassa	Quebec, Canada	250000							3	Temperate	Duchemin et al., (1995; Kelly et al., (1994)
Lake N1	Alaksa	0.5	15					4.96	1.28	Arctic	Kling et al., (1992)
Lake N2-cont	Alaksa	1.8	15						8.5	Arctic	Kling et al., (1992)
Lake N2-fert	Alaksa	1.8	15						3.69	Arctic	Kling et al., (1992)
Lake Windy	Alaksa	12.8	9						3.69	Arctic	Kling et al., (1992)
Lake William	Alaksa	51	12						4.65	Arctic	Kling et al., (1992)
Lake Toolik	Alaksa	150	15					12.1	16.36	Arctic	Kling et al., (1992)

Lake Coleen	Alaksa	190	8					4.97	Arctic	Kling et al., (1992)
Pond 386	Alaksa		10					8.02	Arctic	Kling et al., (1992)
Dam	Alaksa		15					6.9	Arctic	Kling et al., (1992)
Dam	Alaksa		15					8.18	Arctic	Kling et al., (1992)
Hydroreservoirs	China							5.3	Temperate	Li et al., (2015)
Lake Vesijärvi	Enonselkä, Finland	2600		7.4	6.8			3.8	Temperate	López Bellido et al., (2011)
Little Shingboee	North America	2.7						32.98	Temperate	Michmerhuizen et al., (1996)
Glacier pond	North America	7.2						3.32	Temperate	Michmerhuizen et al., (1996)
Hiawatha	North America	21.7						18.54	Temperate	Michmerhuizen et al., (1996)
Crystal	North America	37.9				1.8		0.32	Temperate	Michmerhuizen et al., (1996)
Tofte	North America	50.6						4.3	Temperate	Michmerhuizen et al., (1996)
Jasper	North America	75.8						1.21	Temperate	Michmerhuizen et al., (1996)
Nokomis	North America	80.6						0.16	Temperate	Michmerhuizen et al., (1996)
Harriet	North America	119.5						0.27	Temperate	Michmerhuizen et al., (1996)
Ojibway	North America	152.1						9.75	Temperate	Michmerhuizen et al., (1996)
Allequash	North America	161.2				3.9	29.4	2.05	Temperate	Michmerhuizen et al., (1996)
Calhoun	North America	172						0.41	Temperate	Michmerhuizen et al., (1996)
11th Crow Wing	North America	299.7						6.83	Temperate	Michmerhuizen et al., (1996)

Big Muskellunge	North America	384.1			3.9		22.6	1.66	Temperate	Michmerhuizen et al., (1996)
Trout	North America	1561			2.9		17	0.07	Temperate	Michmerhuizen et al., (1996)
Snowbank	North America	2004						0.29	Temperate	Michmerhuizen et al., (1996)
Minnetonka	North America	5301						0.94	Temperate	Michmerhuizen et al., (1996)
Leech	North America	57340						1.62	Temperate	Michmerhuizen et al., (1996)
Lake Biwa	Eurasia	67400					5.9	3.24	Temperate	Miyajima et al., (1997)
Urban Pond	Linköping, Sweden	0.12		1.2				128.3	Temperate	Natchimuthu et al., (2014)
Streams	Berlin, Germany	0.79				0.489		39	Temperate	Herrero Ortega et al., (2019)
Ponds	Berlin, Germany	2.11						120	Temperate	Herrero Ortega et al., (2019)
Rivers	Berlin, Germany	21.4						20	Temperate	Herrero Ortega et al., (2019)
Lakes	Berlin, Germany	29.7						39	Temperate	Herrero Ortega et al., (2019)
Urban pond	Uppsala,Sweden	0.0029	19.7	8.38		1.11	2.8	0.4	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0091	19.9	8.49		0.53	11.6	1.5	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.011	11	7.7		1.45	7.1	2	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0114	16.5	7.21		1.04	434	4	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0123	19.9	7.66		1.07	45.7	24	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.01297	21.9	8.06		0.587	48.4	12	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0139	20.5	8.04		0.297	9.4	1.4	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.01493	23.5	7.93		0.525	28	14	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0161	16.4	8.19		1.03	3.3	1.4	Temperate	M. Peacock et al., (2019)

Urban pond	Uppsala,Sweden	0.0173	11.6	7.46	3.79	106	75	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0191	15.5	7.74	5.93	47.6	27	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0231	21	7.46	0.989	60.7	12	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0232	17.3	7.44	0.43	64.8	12.5	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0249	24	8.83	1.44	95.1	174	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0257	13.2	7.6	4.25	10.4	3.5	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0291	21.5	7.44	2.09	14.3	60.5	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0353	23.6	7.64	1.02	83	37.5	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0439	23.5	7.96	0.52	21.9	48	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0498	16.2	7.6	0.645	58.6	40	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0537	10.5	7.33	0.355	53.1	2.5	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0544	20.6	7.9	1.42	66	13	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0557	17.5	8.73	0.757	22.1	12	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.058	23.4	7.62	0.691	82.3	13.5	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.067	19.4	7.68	1.08	41.3	12.5	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0783	22.8	7.89	0.574	16.6	5.5	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0799	21.7	7.29	0.417	19.1	9	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0809	19.4	7.24	2.25	800	28	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0893	19.3	7.81	0.915	145	9	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.0908	24.6	7.53	1.24	35	81	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.104	21.4	8.92	0.76	45.5	51	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.118	19.4	7.63	1.01	83.3	20	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.1222	23.6	9.24	3.69	35.9	125	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.1331	17.3	7.06	0.829	552	10	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.2539	19.4	7.82	1.69	160	17.5	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.2588	22.7	8.46	0.5	58.5	12	Temperate	M. Peacock et al., (2019)

Urban pond	Uppsala,Sweden	0.2798	22.2	8.55			0.469	61.2	19	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.4152	21.5	8.31			0.908	105	8	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	0.8325	22.8	7.5			0.779	97.9	149	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	1.138	18	8.19			3.31	45.3	48	Temperate	M. Peacock et al., (2019)
Urban pond	Uppsala,Sweden	1.35	22.6	7.91			1.28	48.1	16	Temperate	M. Peacock et al., (2019)
MVM pond	Sweden		1	7.85	0.9	3.3	0.019	1.2	0.02	Temperate	Peacock et al., (2021)
MVM pond	Sweden		9	7.93	0.9	3.3	0.008	3.2	0.06	Temperate	Peacock et al., (2021)
MVM pond	Sweden		6	8.01	0.9	3.8	0.009	3.9	0.15	Temperate	Peacock et al., (2021)
MVM pond	Sweden		14	7.97	0.9	3.7	0.014	5.4	0	Temperate	Peacock et al., (2021)
MVM pond	Sweden		21	8.11	0.9	4.7	0.009	6.7	0.11	Temperate	Peacock et al., (2021)
MVM pond	Sweden		22	8.35	0.9	3.9	0.011	8.3	0.72	Temperate	Peacock et al., (2021)
MVM pond	Sweden		27	7.92	0.49	3.9	0.006	8.7	2.74	Temperate	Peacock et al., (2021)
MVM pond	Sweden		20	8.14	0.7	5.2	0.007	9.2	0.23	Temperate	Peacock et al., (2021)
MVM pond	Sweden		25	7.91	0.42	4.6	0.008	13.5	2.1	Temperate	Peacock et al., (2021)
Stormwater pond north	Sweden		6	7.33	0.13	5.6	0.015	17.3	0	Temperate	Peacock et al., (2021)
MVM pond	Sweden		19	7.99	0.6	4.3	0.015	20.4	0.23	Temperate	Peacock et al., (2021)
Stormwater pond north	Sweden		22	8.68	0.22	6.6	0.033	25.3	4.93	Temperate	Peacock et al., (2021)
Stormwater pond north	Sweden		6	7.76	0.12	4.7	0.149	32.5	2.88	Temperate	Peacock et al., (2021)
Fembäcke pond S	Sweden		21	7.98	0.09	15	0	35.2	4.48	Temperate	Peacock et al., (2021)
Stormwater pond south	Sweden		22	8.2	0.14	6.4	0.024	36.5	1.92	Temperate	Peacock et al., (2021)
Stormwater pond north	Sweden		14	8.38	0.21	6.2	0.067	37.7	0.09	Temperate	Peacock et al., (2021)
Fembäcke pond S, Sweden	Sweden		7	7.57	0.21	19.3	0.002	38	0.15	Temperate	Peacock et al., (2021)

Stormwater pond north	Sweden	28	8.8	0.18	7.2	0.017	38.9	2.49	Temperate	Peacock et al., (2021)
Trout pond, Sweden	Sweden	1	7.05	0.49	5.7	0.173	41.4	1.16	Temperate	Peacock et al., (2021)
Stormwater pond north	Sweden	22	8.72	0.22	10.1	0	44.1	3.87	Temperate	Peacock et al., (2021)
Stormwater pond south	Sweden	22	9.43	0.11	7	0.005	49.3	1.63	Temperate	Peacock et al., (2021)
Fembäcke pond S, Sweden	Sweden	5	7.27	0.17	22.3	0.002	51.6	0.17	Temperate	Peacock et al., (2021)
Fembäcke pond S, Sweden	Sweden	21	7.07	0.19	27	0	60.3	7.63	Temperate	Peacock et al., (2021)
Stormwater pond north	Sweden	9	7.25	0.31	5.5	0.078	63.9	0.32	Temperate	Peacock et al., (2021)
Stormwater pond south	Sweden	22	8.88	0.24	9.1	0	67.4	42	Temperate	Peacock et al., (2021)
Fembäcke pond N, Sweden	Sweden	28	6.89	0.18	31	0	67.4	37.74	Temperate	Peacock et al., (2021)
Stormwater pond south	Sweden	14	8.85	0.24	6.2	0.008	67.5	5.49	Temperate	Peacock et al., (2021)
Fembäcke pond S, Sweden	Sweden	28	7.22	0.17	29.4	0	69.4	34.37	Temperate	Peacock et al., (2021)
Stormwater pond south	Sweden	26	8.61	0.18	7.3	0.267	69.7	26.09	Temperate	Peacock et al., (2021)
Stormwater pond north	Sweden	20	8.29	0.08	14.3	0	71.3	8.45	Temperate	Peacock et al., (2021)
Stormwater pond south	Sweden	20	7.91	0.14	11.2	0	72.8	56.49	Temperate	Peacock et al., (2021)
Fembäcke pond S	Sweden	19	7.17	0.15	15	0	74.2	61.02	Temperate	Peacock et al., (2021)
Stormwater pond south	Sweden	9	7.66	0.28	4.7	0.128	77.3	2.18	Temperate	Peacock et al., (2021)
Trout pond	Sweden	22	8.09	0.55	11.3	0.011	82.2	8.8	Temperate	Peacock et al., (2021)

Fembäcke pond N	Sweden	19	6.99	0.12	11.4	0	82.4	13.04	Temperate	Peacock et al., (2021)
Fembäcke pond N	Sweden	15	6.82	0.29	24.1	0.002	86.1	0.77	Temperate	Peacock et al., (2021)
Tobo settlement pond	Sweden	0	6.79	0.21	83.3	3.04	88.8	0.59	Temperate	Peacock et al., (2021)
Fembäcke pond S	Sweden	15	7.21	0.25	28.7	0.002	100	7.64	Temperate	Peacock et al., (2021)
Tobo settlement pond	Sweden	8	7.46	0.19	84.3	2	102	0.54	Temperate	Peacock et al., (2021)
Fembäcke pond N	Sweden	7	6.84	0.52	18.4	0.003	107	0.39	Temperate	Peacock et al., (2021)
Stormwater pond south	Sweden	27	8.69	0.19	8	0	112	19.72	Temperate	Peacock et al., (2021)
Trout pond	Sweden	27	8.61	0.15	11.4	0.014	112	3.26	Temperate	Peacock et al., (2021)
Stormwater pond south	Sweden	1	7.39	0.19	4.3	0.717	113	0.13	Temperate	Peacock et al., (2021)
Trout pond	Sweden	22	8.48	0.14	10.2	0.023	113	0.5	Temperate	Peacock et al., (2021)
Fembäcke pond N	Sweden	5	7.02	0.21	22.2	0.004	115	1.16	Temperate	Peacock et al., (2021)
Fembäcke pond S	Sweden	18	7.26	0.32	16.7	0.003	122	13.4	Temperate	Peacock et al., (2021)
Fembäcke pond N	Sweden	21	7.05	0.07	15.7	0	127	14.97	Temperate	Peacock et al., (2021)
Fembäcke pond S	Sweden	28	7.47	0.34	29.1	0.007	127	1.07	Temperate	Peacock et al., (2021)
Tobo settlement pond	Sweden	23	8.78	0.1	82.7	6.89	131	0.85	Temperate	Peacock et al., (2021)
Tobo settlement pond	Sweden	19	8.5	0.03	95	0.789	133	0.18	Temperate	Peacock et al., (2021)
Tobo settlement pond	Sweden	30	7.25	0.18	108	0.004	144	18.21	Temperate	Peacock et al., (2021)
Trout pond	Sweden	28	8.35	0.12	10.6	0.02	146	2.24	Temperate	Peacock et al., (2021)
Tobo settlement pond	Sweden	22	7.56	0.08	87.4	0.008	151		Temperate	Peacock et al., (2021)
Stormwater pond north	Sweden	1	6.95	0.23	4.2	0.556	164	15.6	Temperate	Peacock et al., (2021)

Trout pond	Sweden	9	8.08	0.47	10	0.03	164	0.19	Temperate	Peacock et al., (2021)
Tobo settlement pond	Sweden	20	7.91	0.03	94.2	2.2	171	5.04	Temperate	Peacock et al., (2021)
Fembäcke pond S	Sweden		6.74	0.21	11.7	0.022	183	0	Temperate	Peacock et al., (2021)
Fembäcke pond N	Sweden	21	6.69	0.31	28.1	0.005	188	9.2	Temperate	Peacock et al., (2021)
Trout pond	Sweden	20	8.15	0.39	12	0.03	189	2.07	Temperate	Peacock et al., (2021)
Fembäcke pond N	Sweden	28	7.26	0.29	28.2	0.009	214	2.7	Temperate	Peacock et al., (2021)
Trout pond	Sweden	6	8.31	0.38	10.7	0.003	235	0.24	Temperate	Peacock et al., (2021)
Fizzy pond	Sweden	21	6.74	0.15	13.9	0	252	2.33	Temperate	Peacock et al., (2021)
Stormwater pond north	Sweden	25	8.88	0.19	8.2	0.026	281	18.06	Temperate	Peacock et al., (2021)
Trout pond	Sweden	15	8.23	0.39	10.7	0.002	289	0.83	Temperate	Peacock et al., (2021)
Fizzy pond	Sweden	15	6.86	0.25	13.5	0.001	328	0	Temperate	Peacock et al., (2021)
Fizzy pond	Sweden	21	7.28	0.19	12.4	0	331	0.65	Temperate	Peacock et al., (2021)
Fizzy pond	Sweden	6	6.94	0.21	12.2	0.001	343	7.58	Temperate	Peacock et al., (2021)
Fembäcke pond N	Sweden	18	7.08	0.12	19.9	0.005	387	0.1	Temperate	Peacock et al., (2021)
Tobo settlement pond	Sweden	15	7.59	0.13	90.9	0.005	393	13.41	Temperate	Peacock et al., (2021)
Fizzy pond	Sweden	27	6.61	0.12	16.2	0	407	23.43	Temperate	Peacock et al., (2021)
Fyris pond SLU	Sweden	22	7.13	0.38	12	0	436	12.53	Temperate	Peacock et al., (2021)
Fembäcke pond N	Sweden		6.47	0.52	8.9	0.01	442	1.06	Temperate	Peacock et al., (2021)
Fizzy pond	Sweden	1	6.82	0.48	7.2	0.046	462	0.83	Temperate	Peacock et al., (2021)
Fyris pond SLU	Sweden	22	7.38	0.19	12.4	0.003	486	26.33	Temperate	Peacock et al., (2021)
Fyris pond SLU	Sweden	6	7.64	0.09	15	0.003	491	2.38	Temperate	Peacock et al., (2021)
Fyris pond SLU	Sweden	27	7.35	0.14	13.6	0	513	43.83	Temperate	Peacock et al., (2021)
Trout pond	Sweden	22	8.59	0.32	12.2	0.042	518	0.88	Temperate	Peacock et al., (2021)

Tobo settlement pond	Sweden	7	7.74	0.08	87.5	0.004	527	6.22	Temperate	Peacock et al., (2021)
Fyris pond SLU	Sweden	22	7.46	0.18	13	0.005	623	1109.4	Temperate	Peacock et al., (2021)
Fyris pond SLU	Sweden	18	7.31	0.14	12.9	0	659	88.37	Temperate	Peacock et al., (2021)
Fyris pond SLU	Sweden	28	7.47	0.11	13.5	0	747	46.85	Temperate	Peacock et al., (2021)
Fizzy pond	Sweden	9	6.84	0.52	12.9	0.007	763	46.06	Temperate	Peacock et al., (2021)
Fyris pond SLU	Sweden	15	7.48	0.22	12.2	0.01	890	30.34	Temperate	Peacock et al., (2021)
Fizzy pond	Sweden	20	6.87	0.19	17.6	0	1190	100.07	Temperate	Peacock et al., (2021)
Fyris pond SLU	Sweden	9	7.33	0.42	17.1	0.003	1660	22.8	Temperate	Peacock et al., (2021)
Fyris pond SLU	Sweden	1	6.61	0.34	19	0.246	2050	95.83	Temperate	Peacock et al., (2021)
Fizzy pond	Sweden	22	7.14	0.22	13.8	0	2830	2.48	Temperate	Peacock et al., (2021)
Fizzy pond	Sweden	28	6.93	0.08	13.9	0	3080	2747.69	Temperate	Peacock et al., (2021)
Tobo settlement pond	Sweden	14.2	6.79	0.12				0.292	Temperate	Peacock et al., (2021)
Fembäcke pond N	Sweden	14.5	7.05	0.15				9.45	Temperate	Peacock et al., (2021)
Fyris pond SLU	Sweden	14	7.1	0.51				0.363	Temperate	Peacock et al., (2021)
Fizzy pond	Sweden	13.3	7.35	0.15				0.901	Temperate	Peacock et al., (2021)
Trout pond	Sweden	13.5	7.39	0.6				1.024	Temperate	Peacock et al., (2021)
Fembäcke pond S	Sweden	14.5	7.4	0.16				5.269	Temperate	Peacock et al., (2021)
MVM pond	Sweden	10.8	7.9	0.1				0.068	Temperate	Peacock et al., (2021)
Stormwater pond north	Sweden	18.4	7.94	0.15				0.896	Temperate	Peacock et al., (2021)
Stormwater pond south	Sweden	17.6	8.05	0.33				0.56	Temperate	Peacock et al., (2021)
Tobo settlement pond	Sweden	29						0.24	Temperate	Peacock et al., (2021)
Stormwater pond north	Sweden	19	7.98	0.05	6.3	0	25.2	15.17	Temperate	Peacock et al., (2021)

Ponds	Minnesota, The USA	1		7.5	1	12	0.8	147	219	Temperate	Rabaey and Cotner, (2022)
Crystal Bog	North America	0.54		4.99	2.53	9.3			19.9	Temperate	Riera et al., (1999)
Trout Bog	North America	1.1		4.92	7.9	20.7			27.4	Temperate	Riera et al., (1999)
Crystal Lake	North America	36.65		6.13	20.4	1.9			1.93	Temperate	Riera et al., (1999)
Sparkling Lake	North America	63.9		7.31	20	3.2			2.89	Temperate	Riera et al., (1999)
Lake 227	North America	5.5				9		35	2.04	Temperate	Rudd and Hamilton, (1978)
Lake Okaro	New Zealand	32			12.5	5.2			24.32	Temperate	Santoso et al., (2021)
Lake Rotsee	Lucerne, Switzerland	960	6		9				0.2	Temperate	Schubert et al., (2010)
Lake Rotsee	Lucerne, Switzerland	960	11		9				7	Temperate	Schubert et al., (2010)
Constance	Eurasia	54000				1.5		14.9	0.47	Temperate	Schultz et al., (2008)
AP1, Aquaculture ponds	Kolkata, India	4.402	33	7.78	1.1				595	Tropical	Shaher et al., (2020)
AP2, Aquaculture ponds	Kolkata, India	4.428	33.1	8.04	0.6				145.2	Tropical	Shaher et al., (2020)
Lake Rainbow	North America	1	10.4		1				28.9	Temperate	Smith and Lewis, (1992)
Lake Pass, North America	North America	1.6	6.6		3.5				28.23	Temperate	Smith and Lewis, (1992)
Lake Red Rock, North America	North America	2.1	10		1	10			47.16	Temperate	Smith and Lewis, (1992)
Lake Long, North America	North America	16	9.3		7.5				1.12	Temperate	Smith and Lewis, (1992)
Lake Dillon, North America	North America	1300	12.7		57				21.5	Temperate	Smith and Lewis, (1992)
Oroville reservoi	California, USA	3400	23.8	7.6		1.4			4.2	Temperate	Soumis et al., (2004)
Dworshak reservoir	Idaho, USA	3700	22	8.6		1.61			4.4	Temperate	Soumis et al., (2004)
New Melones reservoir	California, USA	3800	24.3	8.23		1.2			7.1	Temperate	Soumis et al., (2004)

Shasta reservoir	California, USA	7700	23.5	7.6		2.8			9.5	Temperate	Soumis et al., (2004)
Wallula reservoir	Oregon, USA	15700	20.1	8.1		1.3			9	Temperate	Soumis et al., (2004)
F.D. Roosevelt reservoir	Washington, USA	30600	21.7	8.4		2.3			3.2	Temperate	Soumis et al., (2004)
Wintergreen	North America	15			6				673.8	Temperate	Strayer and Tiedje (1978)
Geai	Quebec, Canada	0.008			3.5	9.6		12.8	3.4	Temperate	Thottathil and Prairie, (2021)
Triton	Quebec, Canada	0.017			2.5	5.2		10.4	7.3	Temperate	Thottathil and Prairie, (2021)
Croche	Quebec, Canada	0.063			4.7	5	0.1	5.4	6.5	Temperate	Thottathil and Prairie, (2021)
Cromwell	Quebec, Canada	0.102			3.5	5.5		9.8	11	Temperate	Thottathil and Prairie, (2021)
Arrow-Upper	BC, Canada	52000	12.9	7.9					23	Temperate	Tremblay et al., (2005)
New Melones reservoir	California, USA	3800	24.3	8.23		1.2			7.1	Temperate	Soumis et al., (2004)
Arrow-Narrows	BC, Canada	52000	18.7	8.9					52.9	Temperate	Tremblay et al., (2005)
Arrow-Lower	BC, Canada	52000	19.9	9.1					6.6	Temperate	Tremblay et al., (2005)
Laforge-1	Quebec, Canada	100000	15.1	6.3					27.3	Temperate	Tremblay et al., (2005)
Robertson	Quebec, Canada		13.1	6					6.1	Temperate	Tremblay et al., (2005)
La Grande 1	Quebec, Canada		8.7	6.2					8.8	Temperate	Tremblay et al., (2005)
Baskatong (Mercier)	Quebec, Canada		13.8	6.3					3.2	Temperate	Tremblay et al., (2005)
EOL	Quebec, Canada		16.5	6.3					3.8	Temperate	Tremblay et al., (2005)
La Grande 4	Quebec, Canada		14.4	6.3					10.8	Temperate	Tremblay et al., (2005)
Manic 1	Quebec, Canada		11.9	6.3					11.3	Temperate	Tremblay et al., (2005)
Toulnoustouc	Quebec, Canada		12.4	6.3					0.1	Temperate	Tremblay et al., (2005)
Bersimis	Quebec, Canada		14	6.4					0.1	Temperate	Tremblay et al., (2005)
La Grande 3	Quebec, Canada		11.5	6.4					8.1	Temperate	Tremblay et al., (2005)

Caniapiscau	Quebec, Canada		17.1	6.5			9.8	Temperate	Tremblay et al., (2005)
Gouin	Quebec, Canada		18.3	6.5			2.7	Temperate	Tremblay et al., (2005)
Laforge-2	Quebec, Canada		17.4	6.5			7.5	Temperate	Tremblay et al., (2005)
Outardes 3	Quebec, Canada		14.8	6.5			0.1	Temperate	Tremblay et al., (2005)
Outardes 4	Quebec, Canada		14.6	6.5			0.9	Temperate	Tremblay et al., (2005)
Manic 3	Quebec, Canada		16.4	6.6			1.1	Temperate	Tremblay et al., (2005)
Manic 2	Quebec, Canada		18.3	6.7			6	Temperate	Tremblay et al., (2005)
Manic 5	Quebec, Canada		11	6.7			6.1	Temperate	Tremblay et al., (2005)
Great Fallsanada	Ontario, Canada		22.6	7.5			15.3	Temperate	Tremblay et al., (2005)
Pine Falls	Ontario, Canada		23.2	7.7				Temperate	Tremblay et al., (2005)
Whatshan	BC, Canada		20.8	7.8			5.8	Temperate	Tremblay et al., (2005)
Lac Du Bonnet	Ontario, Canada		23.9	7.9			23.8	Temperate	Tremblay et al., (2005)
Kootenay Lake	BC, Canada		16	8.3			23.5	Temperate	Tremblay et al., (2005)
Duncan Lake	BC, Canada		18.9	8.4			10	Temperate	Tremblay et al., (2005)
Seven Mile	BC, Canada		22.2	8.4			109.5	Temperate	Tremblay et al., (2005)
Waneta	BC, Canada		21.7	8.4			38.8	Temperate	Tremblay et al., (2005)
Day Lake	Wisconsin, USA	2					7	Temperate	St. Louis et al., (2000)
Tigercat Lake	Wisconsin, USA	3					11	Temperate	St. Louis et al., (2000)
Moose Lake	Wisconsin, USA	7					3	Temperate	St. Louis et al., (2000)
Dillon Lake	Wisconsin, USA	13					21	Temperate	St. Louis et al., (2000)
Nelson Lake	Wisconsin, USA	25					6	Temperate	St. Louis et al., (2000)
Chippewa Lake	Wisconsin, USA	61					7	Temperate	St. Louis et al., (2000)
Kasumigaura	Eurasia	16800				4	4.2	Temperate	Utsumi et al., (1998)
Nojiri	Eurasia	440					0.88	Temperate	Utsumi et al., (1998b)
Urban ponds	Gelderland, Netherlands	0.4635	13	7.2	3082	285.2	20	Temperate	van Bergen et al., (2019)

District V, Yangtze River network	Chongqing, China	1730		7.86			12	850	57.4	Subtropical	Wang et al., (2018)
District I, Yangtze River network	Chongqing, China	1960		8.1			3.1	110	4.65	Subtropical	Wang et al., (2018)
District IV, Yangtze River network	Chongqing, China	3100		8.03			9	540	23.3	Subtropical	Wang et al., (2018)
District III, Yangtze River network	Chongqing, China	4380		7.91			8.6	500	17.3	Subtropical	Wang et al., (2018)
District II, Yangtze River network	Chongqing, China	5480		8.02			5	210	10.1	Subtropical	Wang et al., (2018)
Farm Reservoirs	Saskatchewan, Canada	1312		8.75	2.08				7.1	Temperate	Webb et al., (2019)
Small farm reservoirs	Saskatchewan, Canada	0.0158	16	6.95	0.18	4.6	0.417	8.7	6.4	Temperate	Webb et al., (2019)
Small farm reservoirs	Saskatchewan, Canada	0.104	20	8.75	2.1	29.3	2.36	80	51.3	Temperate	Webb et al., (2019)
Small farm reservoirs	Saskatchewan, Canada	0.1312	20	8.75	2.08	31.8	3.08	285	113.9	Temperate	Webb et al., (2019)
Small farm reservoirs	Saskatchewan, Canada	1.39	29.5	10.19	5.1	90.4	1.428	6480	1468	Temperate	Webb et al., (2019)
North Gate Lake	Wisconsin, USA	0.2			8	23.4		17.6	0.37	Temperate	West et al., (2016)
Hummingbird Lake	Wisconsin, USA	0.8			7.6	23		30.7	0.47	Temperate	West et al., (2016)
Cranberry Lake	Wisconsin, USA	1.2			7.9	18.2		33	0.321	Temperate	West et al., (2016)
Paul Lake	Wisconsin, USA	1.4			12	5.1		40.9	0.866	Temperate	West et al., (2016)
Tuesday Lake	Wisconsin, USA	1.6			15	13.4		15.3	0.321	Temperate	West et al., (2016)
Peter Lake	Wisconsin, USA	2.6			18	6.4		12	0.834	Temperate	West et al., (2016)
Raspberry Lake	Wisconsin, USA	4.2			6.1	8.6		23.1	0.433	Temperate	West et al., (2016)
Foggy Lake	Wisconsin, USA	4.4			1.8	11.1		52.6	0.257	Temperate	West et al., (2016)
West Long Lake	Wisconsin, USA	4.9			14	7.4		22.1	0.658	Temperate	West et al., (2016)
Misty Lake	Wisconsin, USA	5.7			13.7	14		21.4	0.369	Temperate	West et al., (2016)
Morris Lake	Wisconsin, USA	5.9			6.7	22.6		36.2	0.225	Temperate	West et al., (2016)
Bergner Lake	Wisconsin, USA	16.2			12	9.4		21.5	0.866	Temperate	West et al., (2016)
Crampton Lake	Wisconsin, USA	25.9			18.5	4.5		11.1	0.417	Temperate	West et al., (2016)

Brown Lake	Wisconsin, U.S.A	29.6			4.9	6.6		86.9	0.192	Temperate	West et al., (2016)
Bay Lake	Wisconsin, U.S.A	69.7			12.2	6.5		22.8	0.257	Temperate	West et al., (2016)
Xiangxi Bay, Three Gorges Reservoir	Xiangxi, China		23.9	7.8					1.944	Subtropical	Xiao et al., (2013)
Xiangxi Bay, Three Gorges Reservoir	Xiangxi, China		28	8.55					2.064	Subtropical	Xiao et al., (2013)
Xiangxi Bay, Three Gorges Reservoir	Xiangxi, China		20	8.8					7.464	Subtropical	Xiao et al., (2013)
Urban Pond	Yichang, China	0.25	29	6.76	1.5		0.067		1.54	Subtropical	Xiao et al., (2014)
Urban Pond	Yichang, China	0.25	20	7.12	1.5		0.067		0.39	Subtropical	Xiao et al., (2014)
Urban Pond	Yichang, China		31.07	6.76					1.54	Subtropical	Xiao et al., (2014)
Urban Pond	Yichang, China		19.8	7.12					0.38	Subtropical	Xiao et al., (2014)
Lake Taihu, Meiliang Bay	Taihu, China	100							1.41	Subtropical	Xiao et al., (2017)
Lake Taihu, Dongtaihu Bay	Taihu, China	131							3.64	Subtropical	Xiao et al., (2017)
Lake Taihu, Gonghu Bay	Taihu, China	215.6							1.03	Subtropical	Xiao et al., (2017)
Lake Taihu, East Zone	Taihu, China	316.4							2.68	Subtropical	Xiao et al., (2017)
Lake Taihu, Northwest Zone	Taihu, China	394.1							3.06	Subtropical	Xiao et al., (2017)
Lake Taihu, Southwest Zone	Taihu, China	443.2							0.63	Subtropical	Xiao et al., (2017)
Lake Taihu, Central Zone	Taihu, China	737.5							0.4	Subtropical	Xiao et al., (2017)
Lake Taihu	Taihu, China	2338							1.48	Subtropical	Xiao et al., (2017)
Three Gorges									7.9	Subtropical	(Yang et al., 2013)

Apx Table C.3 Combined (ebullitive and diffusive) methane emissions and related water chemistry of natural and anthropogenically built waterbodies

Name	Location	Area (ha)	Temperature (°C)	pH	Depth (m)	DOC (mg/l)	TN (mg/l)	TP (ug/l)	Combined emissions (mg/m2/d)	Climate	References
L1 Ladario	South America	6	30	5.9		10.88			262.90	Tropical	Bastviken et al., (2010)
N7a Nhumirim	South America	10	34	10.7		5.76			22.78	Tropical	Bastviken et al., (2010)
L4 Ladario	South America	10	30	6.6		9.73			352.72	Tropical	Bastviken et al., (2010)
L2 Ladario	South America	12	30	6		3.58			118.54	Tropical	Bastviken et al., (2010)
N6b Nhumirim	South America	12	27	8.1		14.05			199.70	Tropical	Bastviken et al., (2010)
N8a Nhumirim	South America	16	29.5	9		3.82			82.93	Tropical	Bastviken et al., (2010)
L3 Ladario	South America	35	30	6.3		1.52			33.52	Tropical	Bastviken et al., (2010)
BB National Park	South America	3630	30.5	6.2		0.74			90.31	Tropical	Bastviken et al., (2010)
TR National Park	South America	7140	31	6		0.75			92.07	Tropical	Bastviken et al., (2010)
Tereza Ladario	South America		24	6.7		1.3			4.01	Tropical	Bastviken et al., (2010)
Presa Ladario	South America		27	6.6		2.85			27.11	Tropical	Bastviken et al., (2010)
Lobo Ladario	South America		24	6.5		6.45			90.14	Tropical	Bastviken et al., (2010)
N14 Nhumirim	South America		29.5	5.7		8.03			331.71	Tropical	Bastviken et al., (2010)
N19a Nhumirim	South America		30	5.3		8.8			187.83	Tropical	Bastviken et al., (2010)
Belém, Ladario	South America		26	6.5					10.27	Tropical	Bastviken et al., (2010)
Bracinho Ladario	South America		26	6.7					249.10	Tropical	Bastviken et al., (2010)
Fontana	Southeastern USA	4300	12.5-26.3		41.4				6	Temperate	Bevelhimer et al., (2016)
Allatoona	Southeastern USA	4900	17.9-26.6		9.2				187	Temperate	Bevelhimer et al., (2016)
Douglas	Southeastern USA	11500	18.1-28.8		11.6				40	Temperate	Bevelhimer et al., (2016)
Watts Bar	Southeastern USA	17600	24.1-27.7		7.9				8	Temperate	Bevelhimer et al., (2016)
Hartwell	Southeastern USA	22600	11.4-28.5		13.9				23	Temperate	Bevelhimer et al., (2016)
Guntersville	Southeastern USA	27900	20.8-26.5		4.5				21	Temperate	Bevelhimer et al., (2016)

Three Gorges	China							6.24	Subtropical	Chen et al., (2011)
Jian River	Beijing, China	0.0086	22	7.8	1.11			23	Temperate	He et al., (2018)
Dam in Chaobai river	China	0.086	28	9.3	2.45			133	Temperate	He et al., (2018)
Jänkäläisenlampi Pond	Finland	1	1.8					7.5	Temperate	Huttunen et al., (2003)
Kotsamolampi Pond	Finland	1	3.2					3.5	Temperate	Huttunen et al., (2003)
Bodaly	Quebec, Canada		16					5	Temperate	Kelly et al., (1994)
Boyd	Quebec, Canada		16.1					2	Temperate	Kelly et al., (1994)
LG2-Bereziuk	Quebec, Canada		12.1		42			10	Temperate	Kelly et al., (1994)
LG2-Ladouceur	Quebec, Canada		15.4					4	Temperate	Kelly et al., (1994)
LG2-Middle	Quebec, Canada		12.5					14	Temperate	Kelly et al., (1994)
LG2-Reservoir	Quebec, Canada		15.8					16	Temperate	Kelly et al., (1994)
LG2-Upstream	Quebec, Canada		11.6		157			4	Temperate	Kelly et al., (1994)
Low	Quebec, Canada		15.6					6	Temperate	Kelly et al., (1994)
Noye	Quebec, Canada		15.9					4	Temperate	Kelly et al., (1994)
Opinaca	Quebec, Canada		15.9		36			15	Temperate	Kelly et al., (1994)
Sakami	Quebec, Canada		15.8		46			3	Temperate	Kelly et al., (1994)
Toto-1	Quebec, Canada		15.3		19			7	Temperate	Kelly et al., (1994)
Toto-2	Quebec, Canada		14.9		19			17	Temperate	Kelly et al., (1994)
Toto-3	Quebec, Canada		15.4		19			50	Temperate	Kelly et al., (1994)
Three Gorges	China							7.92	Subtropical	Lu et al., (2011)
Reservoirs	Mexico City	4.1		7.3	1		11-55.0	4540	Subtropical	Martinez-Cruz et al., (2017)
Ponds	Mexico City	20.82		8.5	0.18			20	Subtropical	Martinez-Cruz et al., (2017)
Rivers	Mexico City	51.7		6.8	1.2		9-35.2	2,400	Subtropical	Martinez-Cruz et al., (2017)

Canals	Mexico City	116.5		7.2				9.6-54.6	800	Subtropical	Martinez-Cruz et al., (2017)
Chinampas	Mexico City	269.48		9	1.27			5.3-6.3	1,200	Subtropical	Martinez-Cruz et al., (2017)
Lakes	Mexico City	554.68		9.3	0.94			8.4-13.2	500	Subtropical	Martinez-Cruz et al., (2017)
Open water in a wetland	Florida, USA	0.95	5-21	7.7					123	Temperate	Morin et al., (2017)
Pond	Linköping, Sweden	0.12			1.2				128	Temperate	Natchimuthu et al. (2014)
Streams	Berlin, Germany	0.79							118	Temperate	Herrero Ortega et al., (2019)
Ponds	Berlin, Germany	2.11							503	Temperate	Herrero Ortega et al., (2019)
Rivers	Berlin, Germany	21.4							123	Temperate	Herrero Ortega et al., (2019)
Lakes	Berlin, Germany	29.7							159	Temperate	Herrero Ortega et al., (2019)
Ponds	Minnesota, USA	1		7.5	1	12	147	0.8	704	Temperate	Rabaey and Cotner, (2022)
Lake Stortjärn	Sweden	4				23	35		2.90	Temperate	Schenk et al., (2021)
Lake Heideweiher	Germany	0.9		4.6		18.1			274.28	Temperate	Schmiedeskamp et al., (2021)
Lake Heideweiher	Germany	0.9		4.6		18.1			121.90	Temperate	Schmiedeskamp et al., (2021)
Lake Windesborn	Germany	1.4		6.9		13.7			105.86	Temperate	Schmiedeskamp et al., (2021)
Lake Windesborn	Germany	1.4		6.9		13.7			91.43	Temperate	Schmiedeskamp et al., (2021)
Ljusvatterntjärn Lake	Sweden	2	18	6.2	9.4	12.2	9.9	0.25	3.95	Temperate	Sieczko et al., (2020)
Erssjön Lake	Sweden	6	7.6	5	4.5	22.5	35	1.1	1.61	Temperate	Sieczko et al., (2020)
Parsen Lake	Sweden	13	10	7	6.7	15.7	17.1	0.62	2.24	Temperate	Sieczko et al., (2020)

Venasjön Lake	Sweden	68	16.8	7.6	11	14.9	48.7	0.78	1.62	Temperate	Sieczko et al., (2020)
North Gate Lake	Wisconsin, USA	0.2			4.1	23.4	17.6		2.37	Temperate	West et al., (2016)
Hummingbird Lake	Wisconsin, USA	0.8			3.4	23	30.7		4.67	Temperate	West et al., (2016)
Bloger Lake	Wisconsin, USA	1.1			2.2	19.5	48.7		15.40	Temperate	West et al., (2016)
Cranberry Lake	Wisconsin, USA	1.2			4.4	18.2	33		1.33	Temperate	West et al., (2016)
Paul Lake	Wisconsin, USA	1.4			3.8	5.1	40.9		75.53	Temperate	West et al., (2016)
Tuesday Lake	Wisconsin, USA	1.6			6.9	13.4	15.3		0.30	Temperate	West et al., (2016)
Peter Lake	Wisconsin, USA	2.6			5.8	6.4	12		27.88	Temperate	West et al., (2016)
Raspberry Lake	Wisconsin, USA	4.2			3	8.6	23.1		0.53	Temperate	West et al., (2016)
Foggy Lake	Wisconsin, USA	4.4			0.7	11.1	52.6		1.17	Temperate	West et al., (2016)
West Long Lake	Wisconsin, USA	4.9			3.9	7.4	22.1		0.67	Temperate	West et al., (2016)
Misty Lake	Wisconsin, USA	5.7			4	14	21.4		0.37	Temperate	West et al., (2016)
Morris Lake	Wisconsin, USA	5.9			2.4	22.6	36.2		52.96	Temperate	West et al., (2016)
Bergner Lake	Wisconsin, USA	16.2			3.7	9.4	21.5		2.69	Temperate	West et al., (2016)
Crampton Lake	Wisconsin, USA	25.9			5	4.5	11		0.42	Temperate	West et al., (2016)
Brown Lake	Wisconsin, USA	29.6			2.7	6.6	86.9		58.26	Temperate	West et al., (2016)
Bay Lake	Wisconsin, USA	69.7			4.2	6.5	22.8		0.90	Temperate	West et al., (2016)
Beaver Pond Harp 4	Ontario, Canada	3.8							37.2	Temperate	Weyhenmeyer, (1999)
CP1 Tai Lake	Suzhou, China	0.09			1.34	127		2140	227	Temperate	Yuan et al., (2021)
CP1 Tai Lake	Suzhou, China	0.71			1.19	146		2010	343	Temperate	Yuan et al., (2021)
CP1 Tai Lake	Suzhou, China	1.17			1.6	123		1640	216	Temperate	Yuan et al., (2021)
Three Gorges	Zigui, China	108400	13-27	7.99	170	7.82	132.3	1.838	3.79	Subtropical	Zhao et al., (2013)
Three Gorges	Xiangxi, China	108400	11.5-30	8.24		6.35	137.2	1.59	9.14	Subtropical	Zhao et al., (2013)
Three Gorges	Badong, China	108400	10.5-26	7.87	110	14.45	161.6	1.888	3.04	Subtropical	Zhao et al., (2013)
Three Gorges	Wanzhou, China	108400	9.6-25.4	7.87	80	8.54	205	1.973	12.83	Subtropical	Zhao et al., (2013)

Apx Table C.4 CSG holding pond information in Queensland

Company name	Pond name	Pond purpose (o=operational; p= proposed)	Pond area (ha)	Pond volume (ML)	Calculated depth (m)	References
QGC	Berwyndale South Pond 4	Produced water (O)	87.4	4,370	5	QGC, (2012a)
QGC	Glen Eden	Produced water (O)	2.3	115	5	QGC, (2012a)
QGC	Kenya Pond	Produced water (O)	40.02	2001	5	QGC, (2012a)
QGC	Rhynie Pond	Produced water (O)	74.38	3719	5	QGC, (2012a)
QGC	Orana 1 Pond	Produced water (O)	103.16	5158	5	QGC, (2012a)
QGC	Orana 5 Pond	Produced water (O)	30.2	2130	7	QGC, (2012a)
QGC	Lauren	Produced water (O)	2.08	104	5	QGC, (2012a)
QGC	David	Produced water (O)	6.68	334	5	QGC, (2012a)
QGC	Janda	Produced water (O)	8.66	433	5	QGC, (2012a)
QGC	Jen 2	Produced water (O)	4.12	206	5	QGC, (2012a)
QGC	Sean	Produced water (O)	3.78	189	5	QGC, (2012a)
QGC	Treated Water Pond	desalinated water (P)	3.4	170	5	QGC, (2012a)
QGC	McNulty	Produced water (P)	4	200	5	QGC, (2012a)
QGC	Ruby – Jo	Produced water (P)	15.2	760	5	QGC, (2012a)
QGC	Kenya East	Produced water (P)	11.94	597	5	QGC, (2012a)
QGC	Celeste	Produced water (P)	6.2	310	5	QGC, (2012a)
QGC	Myrtle	Produced water (P)	3	150	5	QGC, (2012a)
QGC	Glendower	Produced water (P)	9.62	481	5	QGC, (2012a)
QGC	Broadwater	Produced water (P)	7.4	370	5	QGC, (2012a)
QGC	NWTP1 Raw Water	Produced water (O)	12.46	623	5	QGC, (2012a)
QGC	Clarified Water Pond	Produced water (P)	24	1,200	5	QGC, (2012a)
QGC	NWTP 2 Treated Water	desalinated water (P)	6	300	5	QGC, (2012a)

QGC	NWTP 3 RO Reject	Brine (P)	11.5	600	5	QGC, (2012a)
QGC	NWTP 4 Concentrated Brine	Brine (P)	28	1,200	4	QGC, (2012a)
QGC	NWTP 5 Sedimentation	Produced water (P)	4.4	220	5	QGC, (2012a)
QGC	Lawton	Produced water (P)	4	200	5	QGC, (2012a)
QGC	Polaris	Produced water (P)	11.7	583	5	QGC, (2012a)
QGC	Orana 2 (CB pond)	Brine (O)	18.3	1280	7	QGC, (2012b)
QGC	Orana 3 (CB pond)	Brine (O)	20	1040	5	QGC, (2012b)
QGC	Orana 4 (EO reject pond)	Brine (O)	15.6	752	5	QGC, (2012b)
Santos	R-HCS-02 Associated Water Dam	Aggregation (O)	2.339	99.9	4	Santos, (2013, 2012a)
Santos	R-HCS-02 Brine Pond A	Brine (O)	7.6	311.2	4	Santos, (2013, 2012a)
Santos	R-HCS-02 Brine Pond B	Brine (O)	7.315	311.2	4	Santos, (2013, 2012a)
Santos	Angry Jungle CSG water management pond	Aggregation (O)	6.3	181.9	3	Santos, (2013, 2012a)
Santos	ROMA ROP2 Desalinated Water Pond	Permeate water storage (O)	3.1	155	5	Santos, (2013, 2012a)
Santos	Pleasant Hills CSG water management pond	Produced water (O)	4	200	5	Santos, (2013, 2012a)
Santos	Raslie CSG water management pond	Produced water (O)	4	200	5	Santos, (2013, 2012a)
Santos	Grafton Range Dam	Irrigation water storage (O)	n/a	170	n/a	Santos, (2012a)
Santos	Hermitage CSG water management pond	Produced water (O)	4.6	230	5	Santos, (2013, 2012a)
Santos	Ben Bow CSG water management pond	Produced water (O)	3.3	165	5	Santos, (2013, 2012a)
Santos	Treville Downs CSG water management pond	Produced water (O)	2.76	138	5	Santos, (2013, 2012a)
Santos	Mt Hope CSG water management pond	Produced water (O)	2.8	140	5	Santos, (2013, 2012a)

Santos	New Coxon Creek pond	Produced water (O)	3.9	195	5	Santos, (2013, 2012b)
Santos	Pickanjinnie CSG water management pond	Produced water (O)	3.9	195	5	Santos, (2013)
Santos	Pine Ridge CSG water management pond	Produced water (O)	4	195	5	Santos, (2013)
Santos	Coxon Creek East CSG water management pond	Produced water (O)	0.66	33	5	Santos, (2013)
Santos	Washpool Creek CSG water management pond	Produced water (O)	3.7	185	5	Santos, (2013)
Santos	Fairview 77 injection pond	Brine (O)	0.072	3.6	5	Santos, (2013)
Santos	Fairview 82 injection pond	Brine (O)	0.08	4	5	Santos, (2013)
Santos	AWAF2 amended CSG water pond	Produced water (O)	0.34	17	5	Santos, (2013)
Santos	AWAF 2 CSG water management pond	Produced water (O)	0.44	22	5	Santos, (2013)
Santos	Fairview AWAF1 Feed Buffer Dam	Aggregation (O)	4.66	106.44	2	Santos, (2013, 2012a)
Santos	F-HCS-04 Remote Brine Pond A	Brine (O)	8.99	348.1	4	Santos, (2013)
Santos	F-HCS-04 Remote Brine Pond B	Brine (O)	9.15	350	4	Santos, (2013)
Santos	F-HCS-04 Remote Brine Pond C	Brine (O)	8.84	349.49	4	Santos, (2013)
Santos	F-HCS-04 Remote Brine Pond D	Brine (O)	9.01	350.1	4	Santos, (2013)
Santos	F-HCS-04 Associated Water Dam	Aggregation (O)	5.62	204.45	4	Santos, (2013)
Santos	FV ROP2 Desalinated water pond	desalinated water (P)	6.8	340	5	Santos, (2013)
Santos	FV ROP1 CSG water management pond	Produced water (O)	0.76	38	5	Santos, (2013)

Santos	AWAF 3 amended CSG water pond	Amended water (O)	0.86	43	5	Santos, (2013)
Santos	FV ROP1 Desalinated water pond	desalinated water (O)	4.66	233	5	Santos, (2013)
Santos	AWAF3 CSG water management pond	Produced water (O)	0.7	35	5	Santos, (2013)
Santos	Bottle Tree Brine Dam A	Aggregation (O)	7.53	229.9	3	Santos, (2013)
Santos	Mount Kingsley Dam	Aggregation (O)	7.16	225.29	3	Santos, (2013, 2012a)
Santos	Tarcoola CSG water management pond	Produced water (P)	4.8	240	5	Santos, (2013)
Arrow Energy	Tipton West Evaporation Dam 1	Produced water (O)	8	>400	n/a	Mallants et al., (2017)
Arrow Energy	Tipton West Evaporation Dam 2	Produced water (O)	8	>400	n/a	Mallants et al., (2017)
Arrow Energy	Tipton West Pilot Dam	Produced water (O)	8	<400	n/a	Mallants et al., (2017)
Arrow Energy	Tipton West CGPF Dam 1	Oily water discharge (O)	8	400	5	Mallants et al., (2017)
Arrow Energy	Tipton West CGPF Dam 2	Oily water discharge (O)	8	400	5	Mallants et al., (2017)
Arrow Energy	Tipton West Brine Dam	Brine (P)	8	>400	n/a	Mallants et al., (2017)
Arrow Energy	Tipton West Feed Water Dam	Produced water (P)	8	>400	n/a	Mallants et al., (2017)
Arrow Energy	Tipton West Treated Water Dam	Desalinated water (P)	8	>400	n/a	Mallants et al., (2017)
Arrow Energy	Tipton West Utility Dam	RO cleaning wastewater (O)	8	400	5	Mallants et al., (2017)
Arrow Energy	Kogan North Evaporation	Produced water (O)	8	>400	n/a	Mallants et al., (2017)
Arrow Energy	Daandine Feedwater Dam	Produced water (O)	8	>400	n/a	Mallants et al., (2017)
Arrow Energy	Daandine Brine Storage Dam	Brine (O)	8	>400	n/a	Mallants et al., (2017)
Arrow Energy	Daandine Treated Water	Desalinated water (O)	8	<400	n/a	Mallants et al., (2017)
Arrow Energy	Daandine CGPF Dam	Oily water discharge (O)	8	400	5	Mallants et al., (2017)

Arrow Energy	Daandine Utility Dam	RO backwash (O)	8	400	5	Mallants et al., (2017)
Arrow Energy	Stratheden Transfer Dam	Produced water (O)	8	>400	n/a	Mallants et al., (2017)
Arrow Energy	Kenya Brine Pond 1	Brine (P)	22	1500	7	ArrowEnergy, (2018)
Arrow Energy	Kenya Brine Pond 2	Brine (P)	22	1700	8	ArrowEnergy, (2018)
Origin (APLNG)	SGF Pond A Cell 1	Brine (O)	7.5	138	2	APLNG, (2010; Mallants et al., (2017)
Origin (APLNG)	SGF Pond A Cell 2	Produced water (O)	14.4	286	2	Mallants et al., (2017)
Origin (APLNG)	SGF Pond A Cell 3	Brine (O)	22.4	440	2	Mallants et al., (2017)
Origin (APLNG)	SGF Pond A Cell 4	Brine (O)	21.4	340	2	Mallants et al., (2017)
Origin (APLNG)	SGF Pond A Cell 5	Brine (O)	19.8	350	2	Mallants et al., (2017)
Origin (APLNG)	SGF Pond B	Brine (O)	11.6	568	5	Mallants et al., (2017)
Origin (APLNG)	SGF Pond C	Brine (O)	75	1600	2	Mallants et al., (2017)
Origin (APLNG)	Taloona Cell 1	Produced water (O)	4.2	113	3	Mallants et al., (2017)
Origin (APLNG)	Strathblane Pond	Produced water (O)	1	19	2	Mallants et al., (2017)
Origin (APLNG)	Reedy Creek water treatment facility in Combabula Pond 1 -14	Brine (O)	180	6885	4	APLNG, (2010)
Origin (APLNG)	Talinga WTF Feed	Produced water (O)	n/a	436	n/a	Origin, (2017a)
Origin (APLNG)	Condabri WTF Feed	Produced water (O)	n/a	429	n/a	Origin, (2017a)
Origin (APLNG)	Talinga Pond B Cell 1	Brine (O)	n/a	231	n/a	Origin, (2017a)
Origin (APLNG)	Talinga Pond B Cell 2	Brine (O)	n/a	218	n/a	Origin, (2017a)
Origin (APLNG)	Talinga Pond B Cell 3	Brine (O)	n/a	272	n/a	Origin, (2017a)
Origin (APLNG)	Talinga Pond B Cell 4	Brine (O)	n/a	570	n/a	Origin, (2017a)
Origin (APLNG)	Talinga Pond C	Brine (O)	n/a	470	n/a	Origin, (2017a)
Origin (APLNG)	Talinga Pond E	Brine (O)	n/a	7.5	n/a	Origin, (2017a)
Origin (APLNG)	Brine Pond 1	Brine (O)	n/a	306	n/a	Origin, (2017a)
Origin (APLNG)	Brine Pond 2	Brine (O)	n/a	289	n/a	Origin, (2017a)

Origin (APLNG)	Brine Pond 3	Brine (O)	n/a	288	n/a	Origin, (2017a)
Origin (APLNG)	Brine Pond 4	Brine (O)	n/a	247	n/a	Origin, (2017a)
Origin (APLNG)	Brine Pond 5	Brine (O)	n/a	517	n/a	Origin, (2017a)
Origin (APLNG)	Brine Pond 6	Brine (O)	n/a	518	n/a	Origin, (2017a)
Origin (APLNG)	Effluent Brine Pond 1	Brine (O)	n/a	290	n/a	Origin, (2017a)
Origin (APLNG)	ERT Pond	Brine (O)	n/a	53	n/a	Origin, (2017a)
Origin (APLNG)	Brine Pond 7	Brine (P)	n/a	518	n/a	Origin, (2017a)
Origin (APLNG)	Brine Pond 8	Brine (P)	n/a	518	n/a	Origin, (2017a)
Origin (APLNG)	Brine Pond 9	Brine (P)	n/a	518	n/a	Origin, (2017a)
Origin (APLNG)	Brine Pond 10	Brine (P)	n/a	518	n/a	Origin, (2017a)
Origin (APLNG)	Brine Pond 11	Brine (P)	n/a	518	n/a	Origin, (2017a)
Origin (APLNG)	SGWTF Feed Pond-West	Produced water (P)	2	60.1	3	Origin, (2017b)
Origin (APLNG)	SGWTF Feed Pond-East	Produced water (P)	2	60.1	3	Origin, (2017b)
AGL	Produced water pond	Produced water (O)	n/a	375	n/a	AGL, (2014)

References

Aben, R.C.H., Barros, N., Van Donk, E., Frenken, T., Hilt, S., Kazanjian, G., Lamers, L.P.M., Peeters, E.T.H.M., Roelofs, J.G.M., De Senerpont Domis, L.N., Stephan, S., Velthuis, M., Van De Waal, D.B., Wik, M., Thornton, B.F., Wilkinson, J., Delsontro, T., Kosten, S., 2017. Cross continental increase in methane ebullition under climate change. Nat. Commun. 8, 1–8. https://doi.org/10.1038/s41467-017-01535-y

AGL, 2014. Underground Water Impact Report, ATP1056P-Cooper Basin.

Albarède, F., 2009. Geochemistry: An introduction. Cambridge University Press.

Andrews, J.E., Brimblecombe, P., Jickells, T.D., Liss, P.S., 1996. An introduction to Environmental Chemistry. Wiley–Blackwell.

APLNG, 2010. Combabula CSG water management plan.

APPEA, 2018. Queensland Gas: end-to-end water use, supply and management - Report - 2018: the voice of Australia's oil and gas industry.

APPEA, 2017. Coal seam gas water storage ponds : managing risk.

ArrowEnergy, 2018. Surat Gas Project – Brine Pipelines and Ponds.

Atlas, R.M., Bartha, R., 1998. Microbial Ecology: Fundamentals and Applications. Addison-Wesley Publishing Company, London.

Aubinet, M., Vesala, T., Papale, D. (Eds.), 2012. Eddy covariance: a practical guide to measurement and data analysis. Springer Science & Business Media.

Audet, J., Carstensen, M.V., Hoffmann, C.C., Lavaux, L., Thiemer, K., Davidson, T.A., 2020. Greenhouse gas emissions from urban ponds in Denmark. Inl. Waters 10, 373–385. https://doi.org/10.1080/20442041.2020.1730680

Baker-Blocker, A., Donahue, T.M., Mancy, K.H., 1977. Methane flux from wetlands areas. Tellus A Dyn. Meteorol. Oceanogr. 29, 245. https://doi.org/10.3402/tellusa.v29i3.11353

Banks, E.W., Hatch, M., Smith, S., Underschultz, J., Lamontagne, S., Suckow, A., Mallants, D., 2019. Multi-tracer and hydrogeophysical investigation of the hydraulic connectivity between coal seam gas formations, shallow groundwater and stream network in a faulted sedimentary basin. J. Hydrol. 578, 124132. https://doi.org/10.1016/j.jhydrol.2019.124132

Baron, A.A.P., Dyck, L.T., Amjad, H., Bragg, J., Kroft, E., Newson, J., Oleson, K., Casson, N.J., North, R.L., Venkiteswaran, J.J., Whitfield, C.J., 2022. Differences in ebullitive methane release from small, shallow ponds present challenges for scaling. Sci. Total Environ. 802. https://doi.org/10.1016/j.scitotenv.2021.149685

Bastviken, D., Cole, J., Pace, M., Tranvik, L., 2004. Methane emissions from lakes: Dependence of lake characteristics, two regional assessments, and a global estimate. Global Biogeochem. Cycles 18, 1–12. https://doi.org/10.1029/2004GB002238

Bastviken, D., Cole, J.J., Pace, M.L., Van de-Bogert, M.C., 2008. Fates of methane from different lake habitats: Connecting whole-lake budgets and CH4 emissions. J. Geophys. Res. Biogeosciences 113, 1–13. https://doi.org/10.1029/2007JG000608

Bastviken, D., Ejlertsson, J., Tranvik, L., 2002. Measurement of methane oxidation in lakes: A comparison of methods. Environ. Sci. Technol. 36, 3354–3361. https://doi.org/10.1021/es010311p

Bastviken, D., Santoro, A.L., Marotta, H., Pinho, L.Q., Calheiros, D.F., Crill, P., Enrich-Prast, A., 2010. Methane emissions from pantanal, South America, during the low water season: Toward more comprehensive sampling. Environ. Sci. Technol. 44, 5450–5455. https://doi.org/10.1021/es1005048

Bastviken, D., Tranvik, L.J., Downing, J.A., Crill, P.M., Enrich-Prast, A., 2011. Freshwater methane emissions offset the continental carbon sink. Science (80-.). 331, 50. https://doi.org/10.1126/science.1196808

Beaulieu, J.J., DelSontro, T., Downing, J.A., 2019. Eutrophication will increase methane emissions from lakes and impoundments during the 21st century. Nat. Commun. 10, 3–7. https://doi.org/10.1038/s41467-019-09100-5

Bel Hadj Ali, N., Abichou, T., Green, R., 2020. Comparing estimates of fugitive landfill methane emissions using inverse plume modeling obtained with Surface Emission Monitoring (SEM), Drone Emission Monitoring (DEM), and Downwind Plume Emission Monitoring (DWPEM). J. Air Waste Manag. Assoc. 70, 410–424. https://doi.org/10.1080/10962247.2020.1728423

Bevelhimer, M.S., Stewart, A.J., Fortner, A.M., Phillips, J.R., Mosher, J.J., 2016. CO2 is dominant greenhouse gas emitted from six hydropower reservoirs in southeastern United States during peak summer emissions. Water (Switzerland) 8. https://doi.org/10.3390/w8010015

Boehrer, B., Schultze, M., 2008. Stratification of lakes. Rev. Geophys. 46, 1–27. https://doi.org/10.1029/2006RG000210

Boersma, T., Jordaan, S.M., 2017. Whatever happened to the Golden Age of natural gas? Energy Transitions 1, 1–8. https://doi.org/10.1007/s41825-017-0005-4

Boone, D.R., Mathrani, I.M., Liu, Y.T., Menaia, J., Mah, R.A., Boone, J.E., 1993. Isolation and characterization of Methanohalophilus portucalensis sp. nov. and DNA reassociation study of the genus Methanohalophilus. Int. J. Syst. Bacteriol. 43, 431–437.

Boone, D.R., Worakit, S., Mathrani, I.M., Mah, R.A., 1986. Alkaliphilic methanogens from high-pH lake sediments. Syst. Appl. Microbiol. 7, 230–234. https://doi.org/10.1016/S0723-2020(86)80011-X

Boreham, C.J., Boer, R.A. de, 1998. Origin of Gilmore Gas and Oil, Adavale Basin, Central Queensland. APPEA J. 38, 399. https://doi.org/10.1071/aj97019

Boreham, C.J., Korsch, R., Carmichael, D., 1996. The significance of mid-Cretacetous burial and uplift on the maturation and petroleum generation in the Bowen and Surat Basins, eastern Australia, in: Mesozoic Geology of the Eastern Australia Plate Conference.

Bos, G., 2021. Assessment of Microorganisms Found in Coal Seam Water Holding Ponds and in Brine 1–166.

Bräuer, S.L., Cadillo-Quiroz, H., Yashiro, E., Yavitt, J.B., Zinder, S.H., 2006. Isolation of a novel acidiphilic methanogen from an acidic peat bog. Nature 442, 192–194. https://doi.org/10.1038/nature04810

Buan, N.R., 2018. Methanogens: Pushing the boundaries of biology. Emerg. Top. Life Sci. 2, 629–646. https://doi.org/10.1042/ETLS20180031

Bueno de Mesquita, C.P., Wu, D., Tringe, S.G., 2023. Mthyl-based methanogensis: an ecological and genomic review. Physiol. Metab.

Burba, G., 2021. Atmospheric flux measurements, Advances in Spectroscopic Monitoring of the Atmosphere. Elsevier Inc. https://doi.org/10.1016/B978-0-12-815014-6.00004-X

Burke, S.A., Wik, M., Lang, A., Contosta, A.R., Palace, M., Crill, P.M., Varner, R.K., 2019. Long-Term Measurements of Methane Ebullition From Thaw Ponds. J. Geophys. Res. Biogeosciences 124, 2208–2221. https://doi.org/10.1029/2018JG004786

Cadman, S.J., Pain, L., Vuckovic, V., 1998. Bowen and Surat Basins, Clarence-Moreton Basin, Sydney Basin, Gunnedah and other minor onshore basing, Qld, NSW and NT. The department of Primary and Energy, Bureau of Resrouce Science, the Australian Petroleum Accumulations Report 11, pp. 150--154.

Carmichael, D., Boreham, C.J., 1997. Source rock evaluation in the southern Taroom Trough, in: Green, P.M. (Ed.), The Surat and Bowen Basin, South-East Queensland, Queensland Minerals and Energy Review Series. Queensland Department of Mines adn Energy, Brisbane, pp. 193–228.

Casper, P., Maberly, S.C., Hall, G.H., Finlay, B.J., 2000. Fluxes of methane and carbon dioxide from a small productive lake to the atmosphere. Biogeochemistry 49, 1–19. https://doi.org/10.1023/A:1006269900174

Cerling, T.E., Harris, J.M., MacFadden, B.J., Leakey, M.G., Quade, J., Eisenmann, V., Ehieringer, J.R., 1997. Global vegetation change through the Miocene/Pliocene boundary. Nature 389, 153–158. https://doi.org/10.1038/38229

Chau, Y.K., Snodgrass, W.J., Wong, P.T.S., 1977. A sampler for collecting evolved gases from sediment. Water Res. 11, 807–809. https://doi.org/10.1016/0043-1354(77)90051-3

Chen, H., Yuan, X., Chen, Z., Wu, Y., Liu, X., Zhu, D., Wu, N., Zhu, Q., Peng, C., Li, W., 2011. Methane emissions from the surface of the Three Gorges Reservoir. J. Geophys. Res. Atmos. 116, 1–5. https://doi.org/10.1029/2011JD016244

Conrad, R., 2009. The global methane cycle: Recent advances in understanding the microbial processes involved. Environ. Microbiol. Rep. 1, 285–292. https://doi.org/10.1111/j.1758-2229.2009.00038.x

Costa, K.C., Leigh, J.A., 2014. Metabolic versatility in methanogens. Curr. Opin. Biotechnol. 29, 70–75. https://doi.org/10.1016/j.copbio.2014.02.012

Coulthard, T.J., Baird, A.J., Ramirez, J., Waddington, J.M., 2013. Methane Dynamics in Peat: Importance of Shallow Peats and a Novel Reduced-Complexity Approach for Modeling Ebullition. Carbon Cycl. North. Peatlands 173–185. https://doi.org/10.1029/2008GM000811

CRL, 2017. Underground Water Impact Report-Authority To Prospect Atp744P-Galilee Basin.

Dahm, K.G., Guerra, K.L., Xu, P., Drewes, J.E., 2011. Composite geochemical database for coalbed methane produced water quality in the Rocky Mountain region. Environ. Sci. Technol. 45, 7655–7663. https://doi.org/10.1021/es201021n

Danis, C., O'Neill, C., Lackie, M., 2012. Building 3D geological knowledge through regional scale gravity modelling for the Bowen Basin. Explor. Geophys. 43, 8–25. https://doi.org/10.1071/EG11028

Day, S., Amico, M.D., Etheridge, D., Ong, C., Rodger, A., Sherman, B., 2015. Characterisation of Regional Fluxes of Methane in the Surat Basin , Queensland Detection and Flux Determination. Gas Industry Social and Environmental Research Alliance (GISERA).

Day, S., Amico, M.D., Etheridge, D., Ong, C., Rodger, A., Sherman, B., 2013. Characterisation of Regional Fluxes of Methane in the Surat Basin , Queensland Detection and Flux Determination.

Day, S., Dell'Amico, M., Fry, R., Tousi, H., 2014. Field Measurements of Fugitive Emissions from Equipment and Well Casings in Australian Coal Seam Gas Production Facilities. CSIRO, Aust.

Day, S., Marvig, P., White, S., HALLIBURTON, B., 2016a. Methane Emissions from CSG Well Completion Activities: Report for the Department of the Environment and Energy.

Day, S., Tibbett, A., Sestak, S., Knight, C., Marvig, P., Mcgarry, S., Weir, S., Armand, S., van Holst, J., Fry, R., Amico, M.D., Halliburton, B., 2016b. Methane and Volatile Organic Compound Emissions in New South Wales. Report for the New South Wales Environmental Protection Authority. 332.

de Gouw, J.A., Veefkind, J.P., Roosenbrand, E., Dix, B., Lin, J.C., Landgraf, J., Levelt, P.F., 2020. Daily Satellite Observations of Methane from Oil and Gas Production Regions in the United States. Sci. Rep. 10, 1–10. https://doi.org/10.1038/s41598-020-57678-4

Decker, A.D., Klusman, R., Horner, D.M., Anonymous, 1987. Geochemical techniques appllied to the identification and disposal of connate coal water, in: International Coalbed Methane Symposium. pp. 229–242.

Deemer, B.R., Harrison, J.A., Li, S., Beaulieu, J.J., Delsontro, T., Barros, N., Bezerra-Neto, J.F., Powers, S.M., Dos Santos, M.A., Vonk, J.A., 2016. Greenhouse gas emissions from reservoir water surfaces: A new global synthesis. Bioscience 66, 949–964. https://doi.org/10.1093/biosci/biw117

Deemer, B.R., Holgerson, M.A., 2021. Drivers of Methane Flux Differ Between Lakes and Reservoirs, Complicating Global Upscaling Efforts. J. Geophys. Res. Biogeosciences 126, 1–15. https://doi.org/10.1029/2019JG005600

DelSontro, T., Boutet, L., St-Pierre, A., del Giorgio, P.A., Prairie, Y.T., 2016. Methane ebullition and diffusion from northern ponds and lakes regulated by the interaction between temperature and system productivity. Limnol. Oceanogr. 61, S62–S77. https://doi.org/10.1002/lno.10335

Delsontro, T., Kunz, M.J., Kempter, T., Wüest, A., Wehrli, B., Senn, D.B., 2011. Spatial heterogeneity of methane ebullition in a large tropical reservoir. Environ. Sci. Technol. 45, 9866–9873. https://doi.org/10.1021/es2005545

Delsontro, T., McGinnis, D.F., Wehrli, B., Ostrovsky, I., 2015. Size does matter: Importance of large bubbles and small-scale hot spots for methane transport. Environ. Sci. Technol. 49, 1268–1276. https://doi.org/10.1021/es5054286

DISER, 2022. National Greenhouse and Energy Reporting (Measurement) Determination 2008.

DNRM, 2017. Queensland's petroleum and coal seam gas 2015-16.

Duc, N.T., Crill, P., Bastviken, D., 2010. Implications of temperature and sediment characteristics on methane formation and oxidation in lake sediments. Biogeochemistry 100, 185–196. https://doi.org/10.1007/s10533-010-9415-8

Duc, N.T., Silverstein, S., Lundmark, L., Reyier, H., Crill, P., Bastviken, D., 2013. Automated flux chamber for investigating gas flux at water-air interfaces. Environ. Sci. Technol. 47, 968–975. https://doi.org/10.1021/es303848x

Duchemin, E., Lucotte, M., Canuel, R., Chamberland, A., 1995. Production of the greenhouse gases CH4 and CO2 by hydroelectric reservoirs of the boreal region. Global Biogeochem. Cycles 9, 529–540. https://doi.org/10.1029/95GB02202

Duchemin, E., Lucotte, M., Canuel, R., Queiroz, A.G., Almeida, D.C., Pereira, H.C., Jadson, D., 2000. Duchemin et al (2000) Comparison of greenhouse gas emissions.pdf. Verh. Internat. Verein. Limnol. 27, 1391–1395.

Encinas Fernández, J., Peeters, F., Hofmann, H., 2014. Importance of the autumn overturn and anoxic conditions in the hypolimnion for the annual methane emissions from a temperate lake. Environ. Sci. Technol. 48, 7297–7304. https://doi.org/10.1021/es4056164

Erkkilä, K.M., Ojala, A., Bastviken, D., Biermann, T., Heiskanen, J., Lindroth, A., Peltola, O., Rantakari, M., Vesala, T., Mammarella, I., 2018. Methane and carbon dioxide fluxes over a lake: Comparison between eddy covariance, floating chambers and boundary layer method. Biogeosciences 15, 429–445. https://doi.org/10.5194/bg-15-429-2018

Ettwig, K.F., Butler, M.K., Le Paslier, D., Pelletier, E., Mangenot, S., Kuypers, M.M.M., Schreiber, F., Dutilh, B.E., Zedelius, J., De Beer, D., Gloerich, J., Wessels, H.J.C.T., Van Alen, T., Luesken, F., Wu, M.L., Van De Pas-Schoonen, K.T., Op Den Camp, H.J.M., Janssen-Megens, E.M., Francoijs, K.J., Stunnenberg, H., Weissenbach, J., Jetten, M.S.M., Strous, M., 2010. Nitrite-driven anaerobic methane oxidation by oxygenic bacteria. Nature 464, 543–548. https://doi.org/10.1038/nature08883

Evans, P.N., Parks, D.H., Chadwick, G.L., Robbins, S.J., Orphan, V.J., Golding, S.D., Tyson, G.W., 2015. Methane metabolism in the archaeal phylum Bathyarchaeota revealed by genome-centric metagenomics. Science (80-.). 350, 434–438. https://doi.org/10.1126/science.aac7745

Faiz, M., Hendry, P., 2006. Significance of microbial activity in Australian coal bed methane reservoirs - A review. Bull. Can. Pet. Geol. 54, 261–272. https://doi.org/10.2113/gscpgbull.54.3.261

Fallon, R.D., Harrits, S., Hanson, R.S., Brock, T.D., 1980. The role of methane in internal carbon cycling in Lake Mendota during summer stratification. Limnol. Oceanogr. 25, 357–360. https://doi.org/10.4319/lo.1980.25.2.0357

Fell, C., 2014. Water Treatment and Coal Seam Gas.

Feng, H., Guo, J., Peng, C., Ma, X., Kneeshaw, D., Chen, H., Liu, Q., Liu, M., Hu, C., Wang, W., 2023. Global estimates of forest soil methane flux identify a temperate and tropical forest methane sink. Geoderma 429. https://doi.org/10.1016/j.geoderma.2022.116239

Fisher, R.E., France, J.L., Lowry, D., Lanoisellé, M., Brownlow, R., Pyle, J.A., Cain, M., Warwick, N., Skiba, U.M., Drewer, J., Dinsmore, K.J., Leeson, S.R., Bauguitte, S.J.B., Wellpott, A., O'Shea, S.J., Allen, G., Gallagher, M.W., Pitt, J., Percival, C.J., Bower, K., George, C., Hayman, G.D., Aalto, T., Lohila, A., Aurela, M., Laurila, T., Crill, P.M., McCalley, C.K., Nisbet, E.G., 2017. Measurement of the 13C isotopic signature of methane emissions from northern European wetlands. Global Biogeochem. Cycles 31, 605–623. https://doi.org/10.1002/2016GB005504

Freeze, R.A., Cherry, J.A., 1979. Groundwater. Prentice-Hall, Englewood Cliffs, N.J.

Frenzel, P., Thebrath, B., Conrad, R., 1990. Oxidation of methane in the oxic surface layer of a deep lake sediment (Lake Constance). FEMS Microbiol. Lett. 73, 149–158. https://doi.org/10.1016/0378-1097(90)90661-9

Fuchs, A., Selmeczy, G.B., Kasprzak, P., Padisák, J., Casper, P., 2016. Coincidence of sedimentation peaks with diatom blooms, wind, and calcite precipitation measured in high resolution by a multi-trap. Hydrobiologia 763, 329–344. https://doi.org/10.1007/s10750-015-2388-9

Gålfalk, M., Nilsson Påledal, S., Bastviken, D., 2021. Sensitive Drone Mapping of Methane Emissions without the Need for Supplementary Ground-Based Measurements. ACS Earth Sp. Chem. 5, 2668–2676. https://doi.org/10.1021/acsearthspacechem.1c00106

GalileeEnergy, 2022. Underground Water Impact Report, ATP 2019-Galilee Basin.

Gao, M., Vollrath, C., 2023. Global observational coverage of oil and gas methane sources with TROPOMI.

Gao, Y., Liu, X., Yi, N., Wang, Y., Guo, J., Zhang, Z., Yan, S., 2013. Estimation of N2 and N2O ebullition from eutrophic water using an improved bubble trap device. Ecol. Eng. 57, 403–412. https://doi.org/10.1016/j.ecoleng.2013.04.020

Gardner, R.D., Lohman, E., Gerlach, R., Cooksey, K.E., Peyton, B.M., 2013. Comparison of CO2 and bicarbonate as inorganic carbon sources for triacylglycerol and starch accumulation in Chlamydomonas reinhardtii. Biotechnol. Bioeng. 110, 87–96. https://doi.org/10.1002/bit.24592

Ghafari, S., Hasan, M., Aroua, M.K., 2009. Effect of carbon dioxide and bicarbonate as inorganic carbon sources on growth and adaptation of autohydrogenotrophic denitrifying bacteria. J. Hazard. Mater. 162, 1507–1513. https://doi.org/10.1016/j.jhazmat.2008.06.039

Goeckner, A.H., Lusk, M.G., Reisinger, A.J., Hosen, J.D., Smoak, J.M., 2022. Florida's urban stormwater ponds are net sources of carbon to the atmosphere despite increased carbon burial over time. Commun. Earth Environ. 3, 1–8. https://doi.org/10.1038/s43247-022-00384-y

Goodrich, J.P., Varner, R.K., Frolking, S., Duncan, B.N., Crill, P.M., 2011. High-frequency measurements of methane ebullition over a growing season at a temperate peatland site. Geophys. Res. Lett. 38, 1–5. https://doi.org/10.1029/2011GL046915

Gorsky, A.L., Racanelli, G.A., Belvin, A.C., Chambers, R.M., 2019. Greenhouse gas flux from stormwater ponds in southeastern Virginia (USA). Anthropocene 28, 100218. https://doi.org/10.1016/j.ancene.2019.100218

Gravestock, D.I., Hibburt, J.E., Drexel, J.F., 1998. The Petroleum Geology of South Australia, Vol 4: Cooper Basin. SA Department of Primary Industries and Resources, Adelaide.

Grinham, A., Albert, S., Deering, N., Dunbabin, M., Bastviken, D., Sherman, B., Lovelock, C.E., Evans, C.D., 2018. The importance of small artificial water bodies as sources of methane emissions in Queensland, Australia. Hydrol. Earth Syst. Sci. 22, 5281–5298. https://doi.org/10.5194/hess-22-5281-2018

Grinham, A., Dunbabin, M., Gale, D., Udy, J., 2011. Quantification of ebullitive and diffusive methane release to atmosphere from a water storage. Atmos. Environ. 45, 7166–7173. https://doi.org/10.1016/j.atmosenv.2011.09.011

Guanter, L., Irakulis-Loitxate, I., Gorroño, J., Sánchez-García, E., Cusworth, D.H., Varon, D.J., Cogliati, S., Colombo, R., 2021. Mapping methane point emissions with the PRISMA

spaceborne imaging spectrometer. Remote Sens. Environ. 265, 112671. https://doi.org/10.1016/j.rse.2021.112671

Guerrero-Cruz, S., Vaksmaa, A., Horn, M.A., Niemann, H., Pijuan, M., Ho, A., 2021. Methanotrophs: Discoveries, Environmental Relevance, and a Perspective on Current and Future Applications. Front. Microbiol. 12, 1–28. https://doi.org/10.3389/fmicb.2021.678057

Gürsan, C., de Gooyert, V., 2021. The systemic impact of a transition fuel: Does natural gas help or hinder the energy transition? Renew. Sustain. Energy Rev. 138, 110552. https://doi.org/10.1016/j.rser.2020.110552

Hamilton, S.K., Esterle, J.S., Golding, S.D., 2012. Geological interpretation of gas content trends, Walloon Subgroup, eastern Surat Basin, Queensland, Australia. Int. J. Coal Geol. 101, 21–35. https://doi.org/10.1016/j.coal.2012.07.001

Hansell, D.A., 2013. Recalcitrant dissolved organic carbon fractions. Ann. Rev. Mar. Sci. 5, 421–445. https://doi.org/10.1146/annurev-marine-120710-100757

Hao, X., Ruihong, Y., Zhuangzhuang, Z., Zhen, Q., Xixi, L., Tingxi, L., Ruizhong, G., 2021. Greenhouse gas emissions from the water–air interface of a grassland river: a case study of the Xilin River. Sci. Rep. 11, 1–14. https://doi.org/10.1038/s41598-021-81658-x

He, B., He, J., Wang, J., Li, J., Wang, F., 2018. Characteristics of GHG flux from water-air interface along a reclaimed water intake area of the Chaobai River in Shunyi, Beijing. Atmos. Environ. 172, 102–108. https://doi.org/10.1016/j.atmosenv.2017.10.060

Healy, R.W., Bartos, T.T., Rice, C.A., McKinley, M.P., Smith, B.D., 2011. Groundwater chemistry near an impoundment for produced water, Powder River Basin, Wyoming, USA. J. Hydrol. 403, 37–48. https://doi.org/10.1016/j.jhydrol.2011.03.042

Herrero Ortega, S., Romero González-Quijano, C., Casper, P., Singer, G.A., Gessner, M.O., 2019. Methane emissions from contrasting urban freshwaters: Rates, drivers, and a whole-city footprint. Glob. Chang. Biol. 25, 4234–4243. https://doi.org/10.1111/gcb.14799

Holgerson, M.A., Raymond, P.A., 2016. Large contribution to inland water CO2 and CH4 emissions from very small ponds. Nat. Geosci. 9, 222–226. https://doi.org/10.1038/ngeo2654

Holmes, C.D., Prather, M.J., Søvde, O.A., Myhre, G., 2013. Future methane, hydroxyl, and their uncertainties: Key climate and emission parameters for future predictions. Atmos. Chem. Phys. 13, 285–302. https://doi.org/10.5194/acp-13-285-2013

Hounshell, A.G., McClure, R.P., Lofton, M.E., Carey, C.C., 2021. Whole-ecosystem oxygenation experiments reveal substantially greater hypolimnetic methane concentrations in reservoirs during anoxia. Limnol. Oceanogr. Lett. 6, 33–42. https://doi.org/10.1002/lol2.10173

Hug, L.A., Baker, B.J., Anantharaman, K., Brown, C.T., Probst, A.J., Castelle, C.J., Butterfield, C.N., Hernsdorf, A.W., Amano, Y., Ise, K., Suzuki, Y., Dudek, N., Relman, D.A., Finstad, K.M., Amundson, R., Thomas, B.C., Banfield, J.F., 2016. A new view of the tree of life. Nat. Microbiol. 1, 1–6. https://doi.org/10.1038/nmicrobiol.2016.48

Huotari, J., Ojala, A., Peltomaa, E., Nordbo, A., Launiainen, S., Pumpanen, J., Rasilo, T., Hari, P., Vesala, T., 2011. Long-term direct CO2 flux measurements over a boreal lake: Five years of eddy covariance data. Geophys. Res. Lett. 38, 1–5. https://doi.org/10.1029/2011GL048753

Huttunen, J.T., Alm, J., Liikanen, A., Juutinen, S., Larmola, T., Hammar, T., Silvola, J., Martikainen, P.J., 2003. Fluxes of methane, carbon dioxide and nitrous oxide in boreal lakes and potential

anthropogenic effects on the aquatic greenhouse gas emissions. Chemosphere 52, 609–621. https://doi.org/10.1016/S0045-6535(03)00243-1

I' Anson, A., 2013. Coal seam gas in the Galilee Basin, Queensland. The geological and social contexts of resource exploration. University of Sydney.

IPCC, 2018. Global warming of 1.5°C, Special Report on Global Warming of 1.5°C.

Iverach, C.P., Cendón, D.I., Beckmann, S., Hankin, S.I., Manefield, M., Kelly, B.F.J., 2020. Constraining source attribution of methane in an alluvial aquifer with multiple recharge pathways. Sci. Total Environ. 703, 134927. https://doi.org/10.1016/j.scitotenv.2019.134927

Iverach, C.P., Cendón, D.I., Hankin, S.I., Lowry, D., Fisher, R.E., France, J.L., Nisbet, E.G., Baker, A., Kelly, B.F.J., 2015. Assessing Connectivity Between an Overlying Aquifer and a Coal Seam Gas Resource Using Methane Isotopes, Dissolved Organic Carbon and Tritium. Sci. Rep. 5, 1–11. https://doi.org/10.1038/srep15996

Jackson, K.S., Horvath, Z., Hawkins, P.J., 1981. An assessment of the petroleum prospects for the Galilee Basin, Queensland. APEA J. 21, 172–186.

Jacob, D.J., Turner, A.J., Maasakkers, J.D., Sheng, J., Sun, K., Liu, X., Chance, K., Aben, I., McKeever, J., Frankenberg, C., 2016. Satellite observations of atmospheric methane and their value for quantifying methane emissions. Atmos. Chem. Phys. 16, 14371–14396. https://doi.org/10.5194/acp-16-14371-2016

Jacob, D.J., Varon, D.J., Cusworth, D.H., Dennison, P.E., Frankenberg, C., Gautam, R., Guanter, L., Kelley, J., McKeever, J., Ott, L.E., Poulter, B., Qu, Z., Thorpe, A.K., Worden, J.R., Duren, R.M., 2022. Quantifying methane emissions from the global scale down to point sources using satellite observations of atmospheric methane. Atmos. Chem. Phys. 22, 9617–9646. https://doi.org/10.5194/acp-22-9617-2022

Jensen-Schmidt, B., Alexander, E.M., Cotton, T.B., 2006. Structural and tectonic setting, in: Cotton, T.B., Scardigno, M.F., Hibburt, J.E. (Eds.), The Petroleum Geology of South Australia, Volume 2: Eromanga Basin. SA Department of Primary Industries and Resources, Adelaide.

Jervis, D., McKeever, J., Durak, B.O.A., Sloan, J.J., Gains, D., Varon, D.J., Ramier, A., Strupler, M., Tarrant, E., 2021. The GHGSat-D imaging spectrometer. Atmos. Meas. Tech. 14, 2127–2140. https://doi.org/10.5194/amt-14-2127-2021

Joyce, J., Jewell, P.W., 2003. Physical controls on methane ebullition from reservoirs and lakes. Environ. Eng. Geosci. 9, 167–178. https://doi.org/10.2113/9.2.167

Judd, A., 2015. The significance of the 22/4b blow-out site methane emissions in the context of the North Sea. Mar. Pet. Geol. 68, 836–847. https://doi.org/10.1016/j.marpetgeo.2015.07.031

Kankaala, P., Huotari, J., Tulonen, T., Ojala, A., 2013. Lake-size dependent physical forcing drives carbon dioxide and methane effluxes from lakes in a boreal landscape. Limnol. Oceanogr. 58, 1915–1930. https://doi.org/10.4319/lo.2013.58.6.1915

Katayama, T., Yoshioka, H., Mochimaru, H., Meng, X.Y., Muramoto, Y., Usami, J., Ikeda, H., Kamagata, Y., Sakata, S., 2014. Methanohalophilus levihalophilus sp. nov., a slightly halophilic, methylotrophic methanogen isolated from natural gas-bearing deep aquifers, and emended description of the genus Methanohalophilus. Int. J. Syst. Evol. Microbiol. 64, 2089–2093. https://doi.org/10.1099/ijs.0.063677-0

KCB, 2022. South Western Queensland Cooper Basin: Underground waer Impact Report 2022.

Keller, M., Stallard, R.F., 1994. Methane emission by bubbling from Gatun Lake, Panama. J. Geophys. Res. 99, 8307–8319. https://doi.org/10.1029/92JD02170

Kelly, B.F.J., Lu, X., Harris, S.J., Neininger, B.G., Hacker, J.M., Schwietzke, S., Fisher, R.E., France, J.L., Nisbet, E.G., Lowry, D., Van Der Veen, C., Menoud, M., Röckmann, T., 2022. Atmospheric methane isotopes identify inventory knowledge gaps in the Surat Basin, Australia, coal seam gas and agricultural regions. Atmos. Chem. Phys. 22, 15527–15558. https://doi.org/10.5194/acp-22-15527-2022

Kelly, C.A., Rudd, J.W.M., St. Louis, V.L., Moore, T., 1994. Turning attention to reservoir surfaces, a neglected area in greenhouse studies. Eos, Trans. Am. Geophys. Union 75, 332–333. https://doi.org/10.1029/94EO00987

Kirschke, S., Bousquet, P., Ciais, P., Saunois, M., Canadell, J.G., Dlugokencky, E.J., Bergamaschi, P., Bergmann, D., Blake, D.R., Bruhwiler, L., Cameron-Smith, P., Castaldi, S., Chevallier, F., Feng, L., Fraser, A., Heimann, M., Hodson, E.L., Houweling, S., Josse, B., Fraser, P.J., Krummel, P.B., Lamarque, J.F., Langenfelds, R.L., Le Quéré, C., Naik, V., O'doherty, S., Palmer, P.I., Pison, I., Plummer, D., Poulter, B., Prinn, R.G., Rigby, M., Ringeval, B., Santini, M., Schmidt, M., Shindell, D.T., Simpson, I.J., Spahni, R., Steele, L.P., Strode, S.A., Sudo, K., Szopa, S., Van Der Werf, G.R., Voulgarakis, A., Van Weele, M., Weiss, R.F., Williams, J.E., Zeng, G., 2013. Three decades of global methane sources and sinks. Nat. Geosci. 6, 813–823. https://doi.org/10.1038/ngeo1955

Kling, G.W., Kipphut, G.W., Miller, M.C., 1992. The flux of CO2 and CH4 from lakes and rivers in arctic Alaska. Hydrobiologia 240, 23–36. https://doi.org/10.1007/BF00013449

Knoll, A.H., Canfield, D.E., Konhauser, K., 2012. Fundamentals of geobiology. Wiley-Blackwell, Chichester, UK.

Korrensalo, A., Mammarella, I., Alekseychik, P., Vesala, T., Tuittila, E.S., 2022. Plant mediated methane efflux from a boreal peatland complex. Plant Soil 471, 375–392. https://doi.org/10.1007/s11104-021-05180-9

Korsch, R.J., Totterdell, J.M., 2009. Subsidence history and basin phases of the Bowen, Gunnedah and Surat Basins, eastern Australia. Aust. J. Earth Sci. 56, 335–353. https://doi.org/10.1080/08120090802698687

Kuivila, K.M., Murray, J.W., Devol, A.H., Novelli, P.C., 1989. Methane production, sulfate reduction and competition for substrates in the sediments of Lake Washington. Geochim. Cosmochim. Acta 53, 409–416. https://doi.org/10.1016/0016-7037(89)90392-X

Kurth, J.M., Op den Camp, H.J.M., Welte, C.U., 2020. Several ways one goal—methanogenesis from unconventional substrates. Appl. Microbiol. Biotechnol. 104, 6839–6854. https://doi.org/10.1007/s00253-020-10724-7

Laurion, I., Vincent, W.F., MacIntyre, S., Retamal, L., Dupont, C., Francus, P., Pienitz, R., 2010. Variability in greenhouse gas emissions from permafrost thaw ponds. Limnol. Oceanogr. 55, 115–133. https://doi.org/10.4319/lo.2010.55.1.0115

Li, S., Zhang, Q., Bush, R.T., Sullivan, L.A., 2015. Methane and CO2 emissions from China's hydroelectric reservoirs: a new quantitative synthesis. Environ. Sci. Pollut. Res. 22, 5325–5339. https://doi.org/10.1007/s11356-015-4083-9

Liu, L., Wilkinson, J., Koca, K., Buchmann, C., Lorke, A., 2016. The role of sediment structure in gas bubble storage and release. J. Geophys. Res. Biogeosciences 121, 1992–2005. https://doi.org/10.1002/2016JG003456

López Bellido, J., Peltomaa, E., Ojala, A., 2011. An urban boreal lake basin as a source of CO2 and CH4. Environ. Pollut. 159, 1649–1659. https://doi.org/10.1016/j.envpol.2011.02.042

Lovley, D.R., Klug, M.J., 1986. Model for the distribution of sulfate reduction and methanogenesis in freshwater sediments. Geochim. Cosmochim. Acta 50, 11–18. https://doi.org/10.1016/0016-7037(86)90043-8

Lu, F., Yang, L., Wang, X., Duan, X., Mu, Y., Song, W., Zheng, F., Niu, J., Tong, L., Zheng, H., Zhou, Y., Qiu, J., Ouyang, Z., 2011. Preliminary report on methane emissions from the Three Gorges Reservoir in the summer drainage period. J. Environ. Sci. 23, 2029–2033. https://doi.org/10.1016/S1001-0742(10)60668-7

Lu, X., Harris, S.J., Fisher, R.E., France, J.L., Nisbet, E.G., Lowry, D., Röckmann, T., Van Der Veen, C., Menoud, M., Schwietzke, S., Kelly, B.F.J., 2021. Isotopic signatures of major methane sources in the coal seam gas fields and adjacent agricultural districts, Queensland, Australia. Atmos. Chem. Phys. 21, 10527–10555. https://doi.org/10.5194/acp-21-10527-2021

Maeck, A., Delsontro, T., McGinnis, D.F., Fischer, H., Flury, S., Schmidt, M., Fietzek, P., Lorke, A., 2013. Sediment trapping by dams creates methane emission hot spots. Environ. Sci. Technol. 47, 8130–8137. https://doi.org/10.1021/es4003907

Mallants, D., Bekele, E., Schmid, W., Miotlinski, K., Bristow, K., 2017. Literature review : Identification of potential pathways to shallow groundwater of fluids associated with hydraulic fracturing.

Mallett, C.W., Pattison, C., McLennan, T., Balfe, P., Sullivan, D., 1995. Bowen Basin, in: Ward, H.J., Harrington, C.W., Mallett, C.W., Beeston, J.W. (Eds.), Geological of Australian Coal Basins, Special Publicaton No. 1. Geological Society of Australia incorporated Coal Group, Australia, pp. 299–340.

Malyan, S.K., Singh, O., Kumar, A., Anand, G., Singh, R., Singh, S., Yu, Z., Kumar, J., Fagodiya, R.K., Kumar, A., 2022. Greenhouse Gases Trade-Off from Ponds: An Overview of Emission Process and Their Driving Factors. Water (Switzerland) 14, 1–15. https://doi.org/10.3390/w14060970

Martinez-Cruz, K., Gonzalez-Valencia, R., Sepulveda-Jauregui, A., Plascencia-Hernandez, F., Belmonte-Izquierdo, Y., Thalasso, F., 2017. Methane emission from aquatic ecosystems of Mexico City. Aquat. Sci. 79, 159–169. https://doi.org/10.1007/s00027-016-0487-y

Mattson, M.D., Likens, G.E., 1993. Redox reactions of organic matter decomposition in a soft water lake. Biogeochemistry 19, 149–172. https://doi.org/10.1007/BF00000876

Mayr, M.J., Zimmermann, M., Dey, J., Brand, A., Wehrli, B., Bürgmann, H., 2020. Growth and rapid succession of methanotrophs effectively limit methane release during lake overturn. Commun. Biol. 3, 1–9. https://doi.org/10.1038/s42003-020-0838-z

McClure, R.P., Hamre, K.D., Niederlehner, B.R., Munger, Z.W., Chen, S., Lofton, M.E., Schreiber, M.E., Carey, C.C., 2018. Metalimnetic oxygen minima alter the vertical profiles of carbon dioxide and methane in a managed freshwater reservoir. Sci. Total Environ. 636, 610–620. https://doi.org/10.1016/j.scitotenv.2018.04.255

Mendoza-Pascual, M.U., Itoh, M., Aguilar, J.I., Padilla, K.S.A.R., Papa, R.D.S., Okuda, N., 2021. Controlling Factors of Methane in Tropical Lakes of Different Depths. J. Geophys. Res. Biogeosciences 126, 1–15. https://doi.org/10.1029/2020JG005828

Menoud, M., van der Veen, C., Maazallahi, H., Hensen, A., Velzeboer, I., van den Bulk, P., Delre, A., Korben, P., Schwietzke, S., Ardelean, M., Calcan, A., Etiope, G., Baciu, C., Scheutz, C., Schmidt,

M., Röckmann, T., 2022. CH4 isotopic signatures of emissions from oil and gas extraction sites in Romania. Elem. Sci. Anthr. 10, 1–16. https://doi.org/10.1525/elementa.2021.00092

Michmerhuizen, C.M., Striegl, R.G., McDonald, M.E., 1996. Potential methane emission from north-temperate lakes following ice melt. Limnol. Oceanogr. 41, 985–991. https://doi.org/10.4319/lo.1996.41.5.0985

Millar, G.J., Couperthwaite, S.J., Moodliar, C.D., 2016. Strategies for the management and treatment of coal seam gas associated water. Renew. Sustain. Energy Rev. 57, 669–691. https://doi.org/10.1016/j.rser.2015.12.087

Millero, F.J., 1974. The Physical Chemistry of Seawater. Annu. Rev. Earth Planet. Sci. 2, 101–150. https://doi.org/10.1146/annurev.ea.02.050174.000533

Milton, M.J.T., 2005. Remote gas sensing, integrated-path remote sensing, in: Worsfild, P., Townshend, A., Poole, C.E. (Eds.), Encyclopedia of Analytical Science. Elsevier: Oxford, pp. 122–126.

Miyajima, T., Yamada, Y., Wada, E., Nakajima, T., Koitabashi, T., Hanba, Y.T., Yoshii, K., 1997. Distribution of greenhouse gases, nitrite, and δ13C of dissolved inorganic carbon in Lake Biwa: Implications for hypolimnetic metabolism. Biogeochemistry 36, 205–221. https://doi.org/10.1023/A:1005702707183

Mokashi, K., Shetty, V., George, S.A., Sibi, G., 2016. Sodium Bicarbonate as Inorganic Carbon Source for Higher Biomass and Lipid Production Integrated Carbon Capture in Chlorella vulgaris. Achiev. Life Sci. 10, 111–117. https://doi.org/10.1016/j.als.2016.05.011

Morin, T.H., Bohrer, G., Stefanik, K.C., Rey-Sanchez, A.C., Matheny, A.M., Mitsch, W.J., 2017. Combining eddy-covariance and chamber measurements to determine the methane budget from a small, heterogeneous urban floodplain wetland park. Agric. For. Meteorol. 237–238, 160–170. https://doi.org/10.1016/j.agrformet.2017.01.022

Morris, J.R., Martin, M.A., 2017. Coal architecture, high-resolution correlation and connectivity: New insights from the walloon subgroup in the Surat basin of SE Queensland, Australia. Pet. Geosci. 23, 251–261. https://doi.org/10.1144/petgeo2016-028

Myers, M., White, C., Pejcic, B., Feitz, A., Roberts, J., Oh, Y.Y., Xu, L., Ricard, L., Michael, K., Avijegon, A., Rachakonda, P.K., Woltering, M., Larcher, A., Stalker, L., Hortle, A., 2020. CSIRO In-Situ Lab: A multi-pronged approach to surface gas and groundwater monitoring at geological CO2 storage sites. Chem. Geol. 545, 119642. https://doi.org/10.1016/j.chemgeo.2020.119642

Natchimuthu, S., Panneer Selvam, B., Bastviken, D., 2014. Influence of weather variables on methane and carbon dioxide flux from a shallow pond. Biogeochemistry 119, 403–413. https://doi.org/10.1007/s10533-014-9976-z

Nauer, P.A., Chiri, E., Jirapanjawat, T., Greening, C., Cook, P.L.M., 2021. Technical note: Inexpensive modification of Exetainers for the reliable storage of trace-level hydrogen and carbon monoxide gas samples. Biogeosciences 18, 729–737. https://doi.org/10.5194/bg-18-729-2021

Navi, M., Skelly, C., Taulis, M., Nasiri, S., 2015. Coal seam gas water: Potential hazards and exposure pathways in Queensland. Int. J. Environ. Health Res. 25, 162–183. https://doi.org/10.1080/09603123.2014.915018

Neininger, B.G., Kelly, B.F.J., Hacker, J.M., Lu, X., Schwietzke, S., 2021. Coal seam gas industry methane emissions in the Surat Basin, Australia: Comparing airborne measurements with

inventories. Philos. Trans. R. Soc. A Math. Phys. Eng. Sci. 379. https://doi.org/10.1098/rsta.2020.0458

Nghiem, L.D., Elters, C., Simon, A., Tatsuya, T., Price, W., 2015. Coal seam gas produced water treatment by ultrafiltration, reverse osmosis and multi-effect distillation: A pilot study. Sep. Purif. Technol. 146, 94–100. https://doi.org/10.1016/j.seppur.2015.03.022

Nghiem, L.D., Naj Aziz, T.R., Porter, I., Regmi, G., 2011. Treatment of coal seam gas produced water for beneficial use in Ausrtalia: A review of best proctices. Desalin. Water Treat. 32, 316–323.

Obraztsova, A.Y., Shipin, O.V., Bezrukova, L.V., Belyaev, S.S., 1987. Properties of the cocoid methylotrophic methanogen, Methanococcoides euhalobius sp. nov. Microbiology 56, 523–527.

Offre, P., Spang, A., Schleper, C., 2013. Archaea in biogeochemical cycles. Annu. Rev. Microbiol. 67, 437–457. https://doi.org/10.1146/annurev-micro-092412-155614

OGIA, 2021. Underground Water Impact Report for the Surat Cumulative Management Area, Queensland Water Commission.

Ong, C., Day, S., Halliburton, B., Marvig, P., White, S., 2017. Regional Methane Emissions in NSW CSG Basins.

Ong, C., Day, S., Maney, B., Mainson, M., Myers, M., Down, D., 2019. Final Report for GISERA Project G5 - Pre-Exploration Measurement and Monitoring of Background Landscape Methane Concentrations and Fluxes in the Beetaloo Sub-Basin, Northern Territory, CSIRO Australia,.

Op den Camp, H.J.M., Islam, T., Stott, M.B., Harhangi, H.R., Hynes, A., Schouten, S., Jetten, M.S.M., Birkeland, N.K., Pol, A., Dunfield, P.F., 2009. Environmental, genomic and taxonomic perspectives on methanotrophic Verrucomicrobia. Environ. Microbiol. Rep. 1, 293–306. https://doi.org/10.1111/j.1758-2229.2009.00022.x

Origin, 2017a. Talinga-Condabri Integrated CSG Water Management Plan.

Origin, 2017b. Spring Gully coal seam gas water management plan.

Orr, B.J., He, Y., 2014. Cavity-based absorption spectroscopy techniques, Laser Spectroscopy for Sensing: Fundamentals, Techniques and Applications. https://doi.org/10.1533/9780857098733.2.167

Pajares, S., Ramos, R., 2019. Processes and Microorganisms Involved in the Marine Nitrogen Cycle: Knowledge and Gaps. Front. Mar. Sci. 6. https://doi.org/10.3389/fmars.2019.00739

Pandey, S., Gautam, R., Houweling, S., Van Der Gon, H.D., Sadavarte, P., Borsdorff, T., Hasekamp, O., Landgraf, J., Tol, P., Van Kempen, T., Hoogeveen, R., Van Hees, R., Hamburg, S.P., Maasakkers, J.D., Aben, I., 2019. Satellite observations reveal extreme methane leakage from a natural gas well blowout. Proc. Natl. Acad. Sci. U. S. A. 116, 26376–26381. https://doi.org/10.1073/pnas.1908712116

Paris, J.D., Riandet, A., Bourtsoukidis, E., Delmotte, M., Berchet, A., Williams, J., Ernle, L., Tadic, I., Harder, H., Lelieveld, J., 2021. Shipborne measurements of methane and carbon dioxide in the Middle East and Mediterranean areas and the contribution from oil and gas emissions. Atmos. Chem. Phys. 21, 12443–12462. https://doi.org/10.5194/acp-21-12443-2021

Paterek, J.R., Smith, P.H., 1988. Methanohalophilus mahii [gen. nov., sp. nov.] a methylotrophic halophilic methanogen. Int. J. Syst. Bacteriol. 38, 122–123.

Peacock, M., Audet, J., Bastviken, D., Cook, S., Evans, C.D., Grinham, A., Holgerson, M.A., Högbom, L., Pickard, A.E., Zieliński, P., Futter, M.N., 2021. Small artificial waterbodies are widespread and persistent emitters of methane and carbon dioxide. Glob. Chang. Biol. 27, 5109–5123. https://doi.org/10.1111/gcb.15762

Peacock, M., Audet, J., Jordan, S., Smeds, J., Wallin, M.B., 2019. Greenhouse gas emissions from urban ponds are driven by nutrient status and hydrology. Ecosphere 10. https://doi.org/10.1002/ecs2.2643

Peacock, Mike, Audet, J., Jordan, S., Smeds, J., Wallin, M.B., 2019. Greenhouse gas emissions from urban ponds are driven by nutrient status and hydrology. Ecosphere 10. https://doi.org/10.1002/ecs2.2643

Pearce, J.K., Golding, S.D., Baublys, K., Hofmann, H., Gargiulo, G., Herbert, S.J., Hayes, P., 2023. Methane in aquifers and alluvium overlying a coal seam gas region: Gas concentrations and isotopic differentiation. Sci. Total Environ. 861, 160639. https://doi.org/10.1016/j.scitotenv.2022.160639

Peltola, O., Mammarella, I., Haapanala, S., Burba, G., Vesala, T., 2013. Field intercomparison of four methane gas analyzers suitable for eddy covariance flux measurements. Biogeosciences 10, 3749–3765. https://doi.org/10.5194/bg-10-3749-2013

Plumlee, M.H., Debroux, J.F., Taffler, D., Graydon, J.W., Mayer, X., Dahm, K.G., Hancock, N.T., Guerra, K.L., Xu, P., Drewes, J.E., Cath, T.Y., 2014. Coalbed methane produced water screening tool for treatment technology and beneficial use. J. Unconv. Oil Gas Resour. 5, 22–34. https://doi.org/10.1016/j.juogr.2013.12.002

QGC, 2012a. Section 14 Ensuring responsible CSG water management and beneficial use, in: Stage 2 CSG Water Monitoring and Management Plan. pp. 196–241.

QGC, 2012b. Section 15 Brine Management Strategy, in: Stage 2 CSG Water Monitoring and Management Plan.

Rabaey, J., Cotner, J., 2022. Pond greenhouse gas emissions controlled by duckweed coverage. Front. Environ. Sci. 10. https://doi.org/10.3389/fenvs.2022.889289

RESTEK, 2017. A Guide to Whole Air Canister Sampling: Equipment Needed and Practical Techniques for Collecting Air Samples 1–20.

Riera, J.L., Schindler, J.E., Kratz, T.K., 1999. Seasonal dynamics of carbon dioxide and methane in two clear-water lakes and two bog lakes in northern Wisconsin, U.S.A. Can. J. Fish. Aquat. Sci. 56, 265–274. https://doi.org/10.1139/f98-182

Robertson, J., 2018. Redefining csg “waste” water: New opportunities for managed aquifer recharge. Environ. Plan. Law J. 35, 188–211.

Robison, A.L., Wollheim, W.M., Perryman, C.R., Cotter, A.R., Mackay, J.E., Varner, R.K., Clarizia, P., Ernakovich, J.G., 2022. Dominance of Diffusive Methane Emissions From Lowland Headwater Streams Promotes Oxidation and Isotopic Enrichment. Front. Environ. Sci. 9, 1–14. https://doi.org/10.3389/fenvs.2021.791305

Rosa, L.P., Dos Santos, M.A., Matvienko, B., Sikar, E., Lourenço, R.S.M., Menezes, C.F., 2003. Biogenic gas production from major Amazon reservoirs, Brazil. Hydrol. Process. 17, 1443–1450. https://doi.org/10.1002/hyp.1295

Rosentreter, J.A., Borges, A. V., Deemer, B.R., Holgerson, M.A., Liu, S., Song, C., Melack, J., Raymond, P.A., Duarte, C.M., Allen, G.H., Olefeldt, D., Poulter, B., Battin, T.I., Eyre, B.D., 2021.

Half of global methane emissions come from highly variable aquatic ecosystem sources. Nat. Geosci. 14, 225–230. https://doi.org/10.1038/s41561-021-00715-2

Rudd, J.W.M., Hamilton, R.D., 1978. Methane cycling in a cutrophic shield lake and its effects on whole lake metabolism. Limnol. Oceanogr. 23, 337–348. https://doi.org/10.4319/lo.1978.23.2.0337

Sadavarte, P., Pandey, S., Maasakkers, J.D., Lorente, A., Borsdorff, T., Denier van der Gon, H., Houweling, S., Aben, I., 2021. Methane Emissions from Superemitting Coal Mines in Australia Quantified Using TROPOMI Satellite Observations. Environ. Sci. Technol. 55, 16573–16580. https://doi.org/10.1021/acs.est.1c03976

Salmachi, A., Rajabi, M., Wainman, C., Mackie, S., McCabe, P., Camac, B., Clarkson, C., 2021. History, geology, in situ stress pattern, gas content and permeability of coal seam gas basins in Australia: A review. Energies 14, 1–37. https://doi.org/10.3390/en14092651

Sanches, L.F., Guenet, B., Marinho, C.C., Barros, N., de Assis Esteves, F., 2019. Global regulation of methane emission from natural lakes. Sci. Rep. 9, 1–10. https://doi.org/10.1038/s41598-018-36519-5

Santos, 2013. CSG Water Monitoring and Management Plan, Summary Plan -Stage 2 Revision 2.

Santos, 2012a. Coal Seam Gas Waste Management Plan GLNG Gas Field Development Project EPBC (2012/6615).

Santos, 2012b. Coal Seam Gas Waste Management Plan Project Stage 2 -Maisey (PL1021) EPBC (2012/6615).

Santoso, A.B., Hamilton, D.P., Schipper, L.A., Ostrovsky, I.S., Hendy, C.H., 2021. High contribution of methane in greenhouse gas emissions from a eutrophic lake: a mass balance synthesis. New Zeal. J. Mar. Freshw. Res. 55, 411–430. https://doi.org/10.1080/00288330.2020.1798476

Saunois, M., Stavert, A.R., Poulter, B., Bousquet, P., Canadell, J.G., Jackson, R.B., Raymond, P.A., Al., E., 2020. Overview of Greenhouse Gases | US EPA. Earth Syst. Sci. Data 12, 1561–1623.

Scheibner, E., 1999. The geological evolution of New South Wales- A brief review. Geological Survey of New South Wales, Sydney.

Schenk, J., Sawakuchi, H.O., Sieczko, A.K., Pajala, G., Rudberg, D., Hagberg, E., Fors, K., Laudon, H., Karlsson, J., Bastviken, D., 2021. Methane in Lakes: Variability in Stable Carbon Isotopic Composition and the Potential Importance of Groundwater Input. Front. Earth Sci. 9, 1–15. https://doi.org/10.3389/feart.2021.722215

Schmiedeskamp, M., Praetzel, L.S.E., Bastviken, D., Knorr, K.H., 2021. Whole-lake methane emissions from two temperate shallow lakes with fluctuating water levels: Relevance of spatiotemporal patterns. Limnol. Oceanogr. 66, 2455–2469. https://doi.org/10.1002/lno.11764

Schrier-Uijl, A.P., Veraart, A.J., Leffelaar, P.A., Berendse, F., Veenendaal, E.M., 2011. Release of CO2 and CH4 from lakes and drainage ditches in temperate wetlands. Biogeochemistry 102, 265–279. https://doi.org/10.1007/s10533-010-9440-7

Schubert, C.J., Diem, T., Eugster, W., 2012. Methane emissions from a small wind shielded lake determined by eddy covariance, flux chambers, anchored funnels, and boundary model calculations: A comparison. Environ. Sci. Technol. 46, 4515–4522. https://doi.org/10.1021/es203465x

Schubert, C.J., Lucas, F.S., Durisch-Kaiser, E., Stierli, R., Diem, T., Scheidegger, O., Vazquez, F., Müller, B., 2010. Oxidation and emission of methane in a monomictic lake (Rotsee, Switzerland). Aquat. Sci. 72, 455–466. https://doi.org/10.1007/s00027-010-0148-5

Schultz, M.G., Heil, A., Hoelzemann, J.J., Spessa, A., Thonicke, K., Goldammer, J.G., Held, A.C., Pereira, J.M.C., van Het Bolscher, M., 2008. Global wildland fire emissions from 1960 to 2000. Global Biogeochem. Cycles 22, 1–17. https://doi.org/10.1029/2007GB003031

Schwarz, J.I.K., Eckert, W., Conrad, R., 2008. Response of the methanogenic microbial community of a profundal lake sediment (Lake Kinneret, Israel) to algal deposition. Limnol. Oceanogr. 53, 113–121. https://doi.org/10.4319/lo.2008.53.1.0113

Scott, S.G., Beeston, J.W., Carr, A.F., 1995. Galilee Basin, in: Ward, .CR., Harrington, H.J., Mallett, C.W., Beeston, J.W. (Eds.), Geology of Australian Coal Basins. Geological Society of Australia – Coal Geology Group, pp. 341–353.

Sestak, S., Armand, S., Vergara, T., 2017. Sapmling for seeps, flux chambers, desorption canisters and air samples for stable isotope.

Shaher, S., Chanda, A., Das, S., Das, I., Giri, S., Samanta, S., Hazra, S., Mukherjee, A.D., 2020. Summer methane emissions from sewage water–fed tropical shallow aquaculture ponds characterized by different water depths. Environ. Sci. Pollut. Res. 27, 18182–18195. https://doi.org/10.1007/s11356-020-08296-0

Shao, L., Mei, J., Chen, J., Tan, T., Wang, G., Liu, K., Gao, X., 2022. Recent advances and applications of off-axis integrated cavity output spectroscopy. Microw. Opt. Technol. Lett. 1–17. https://doi.org/10.1002/mop.33220

Shaw, R.D., Korsch, R.J., Boreham, C.J., Totterdell, J.M., Lelbach, C., Nicoll, M.G., 2000. Evaluation of the undiscovered hydrocarbon resources of the Bowen and Surat Basins, southern Queensland. AGSO J. Aust. Geol. Geophys. 17, 43–65.

Sieczko, A.K., Thanh Duc, N., Schenk, J., Pajala, G., Rudberg, D., Sawakuchi, H.O., Bastviken, D., 2020. Diel variability of methane emissions from lakes. Proc. Natl. Acad. Sci. U. S. A. 117, 21488–21494. https://doi.org/10.1073/pnas.2006024117

Slonczewski, J.L., Foster, J.W., 2017. Microbiology: an evolving science, Fourth edi. ed. New York: W.W. Norton & Comany.

Smith, L.K., Lewis, W.M., 1992. wetland sites in the Rockies of accumulation was as high one of the lakes . For wetlands near the. Global Biogeochem. Cycles 6, 323–338.

Snow, N.H., Slack, G.C., 2002. Head-space analysis in modern gas chromatography. TrAC - Trends Anal. Chem. 21, 608–617. https://doi.org/10.1016/S0165-9936(02)00802-6

Sobek, S., Delsontro, T., Wongfun, N., Wehrli, B., 2012. Extreme organic carbon burial fuels intense methane bubbling in a temperate reservoir. Geophys. Res. Lett. 39, 2–5. https://doi.org/10.1029/2011GL050144

Sorokin, Di.Y., Makarova, K.S., Abbas, B., Ferrer, M., Golyshin, P.N., Galinski, E.A., Ciordia, S., Mena, M.C., Merkel, A.Y., Wolf, Y.I., Van Loosdrecht, M.C.M., Koonin, E. V., 2017. Discovery of extremely halophilic, methyl-reducing euryarchaea provides insights into the evolutionary origin of methanogenesis. Nat. Microbiol. 2. https://doi.org/10.1038/nmicrobiol.2017.81

Soumis, N., Duchemin, É., Canuel, R., Lucotte, M., 2004. Greenhouse gas emissions from reservoirs of the western United States. Global Biogeochem. Cycles 18, 1–11. https://doi.org/10.1029/2003GB002197

St. Louis, V.L., Kelly, C.A., Duchemin, É., Rudd, J.W.M., Rosenberg, D.M., 2000. Reservoir surfaces as sources of greenhouse gases to the atmosphere: A global estimate. Bioscience 50, 766–775. https://doi.org/10.1641/0006-3568(2000)050[0766:RSASOG]2.0.CO;2

Stein, L.Y., Roy, R., Dunfield, P.. F., 2012. Aerobic Methanotrophy and Nitrification: Processes and Connections. eLS. https://doi.org/10.1002/9780470015902.a0022213

Strayer, R.F., Tiedje, J.M., 1978. In situ methane production in a small, hypereutrophic, hard-water lake: Loss of methane from sediments by vertical diffusion and ebullition. Limnol. Oceanogr. 23, 1201–1206. https://doi.org/10.4319/lo.1978.23.6.1201

Stumm, W., Morgan, J.J., 1996. Aquatic chemistry, Third Edit. ed. Wiley-Interscience, New York.

Taulis, M., Milke, M., 2007. Coal seam gas water from Maramarua, New Zealand: Characterisation and comparison to United States analogues. J. Hydrol. New Zeal. 46, 1–17.

Thanh Duc, N., Silverstein, S., Wik, M., Crill, P., Bastviken, D., K. Varner, R., 2020. Technical note: Greenhouse gas flux studies: An automated online system for gas emission measurements in aquatic environments. Hydrol. Earth Syst. Sci. 24, 3417–3430. https://doi.org/10.5194/hess-24-3417-2020

Thornton, B.F., Wik, M., Crill, P.M., 2015. Climate-forced changes in available energy and methane bubbling from subarctic lakes. Geophys. Res. Lett. 42, 1936–1942. https://doi.org/10.1002/2015GL063189

Thottathil, S.D., Prairie, Y.T., 2021. Coupling of stable carbon isotopic signature of methane and ebullitive fluxes in northern temperate lakes. Sci. Total Environ. 777, 146117. https://doi.org/10.1016/j.scitotenv.2021.146117

Towler, B., Firouzi, M., Underschultz, J., Rifkin, W., Garnett, A., Schultz, H., Esterle, J., Tyson, S., Witt, K., 2016. An overview of the coal seam gas developments in Queensland. J. Nat. Gas Sci. Eng. 31, 249–271. https://doi.org/10.1016/j.jngse.2016.02.040

Tremblay, A., Varfalvy, L., Roehm, C., Garneau, M., 2005. Greenhouse Gas Emissions-Fluxes and Processes. Springer Berlin Heidelberg New York.

Troup, A., Longdon, N., Gorton, J., 2018. Petroleum Plays of the Bowen and Surat basins. ASEG Ext. Abstr. 2018, 1–10. https://doi.org/10.1071/aseg2018abm3_1c

Troup, A., Talebi, B., 2019. Adavale Basin petroleum plays. APPEA J. 59, 958. https://doi.org/10.1071/aj18083

Turner, I.L., Acworth, R.I., 2004. Field measurements of beachface salinity structure using cross-borehole resistivity imaging. J. Coast. Res. 20, 753–760. https://doi.org/10.2112/1551-5036(2004)20[753:fmobss]2.0.co;2

Utsumi, M., Nojiri, Y., Nakamura, T., Nozawa, T., Otsuki, A., Seki, H., 1998a. Oxidation of dissolved methane in a eutrophic, shallow lake: Lake Kasumigaura, Japan. Limnol. Oceanogr. 43, 471–480. https://doi.org/10.4319/lo.1998.43.3.0471

Utsumi, M., Nojiri, Y., Nakamura, T., Nozawa, T., Otsuki, A., Takamura, N., Watanabe, M., Seki, H., 1998b. Dynamics of dissolved methane and methane oxidation in dimictic Lake Nojiri during winter. Limnol. Oceanogr. 43, 10–17. https://doi.org/10.4319/lo.1998.43.1.0010

van Bergen, T.J.H.M., Barros, N., Mendonça, R., Aben, R.C.H., Althuizen, I.H.J., Huszar, V., Lamers, L.P.M., Lürling, M., Roland, F., Kosten, S., 2019. Seasonal and diel variation in greenhouse gas emissions from an urban pond and its major drivers. Limnol. Oceanogr. 64, 2129–2139. https://doi.org/10.1002/lno.11173

Van Holst, J., Stalker, L., Yen, L., Stephen, S., 2010. GAS STABLE ISOTOPE ANALYSIS – SAMPLE CONTAINMENT AND δ C STABILITY, in: Chen, J.H., Hope, J.M., Grosjean, E., Boreham, C.J., Brocks, J.J. (Eds.), 16th Australian Organic Geochemistry Conference:Program and Abstracts. Geoscience Australia, pp. 102–103.

Varadharajan, C., Hermosillo, R., Hemond, H.F., 2010. A low-cost automated trap to measure bubbling gas fluxes. Limnol. Oceanogr. Methods 8, 363–375. https://doi.org/10.4319/lom.2010.8.363

Varon, D.J., Jervis, D., McKeever, J., Spence, ., Gains, D., Jacob, D.J., 2021. High-frequency monitoring of anomalous methane point sources with multispectral Sentinel-2 satellite observations. Atmos. Meas. Tech. 14, 2771–2785. https://doi.org/10.5194/amt-14-2771-2021

Vesala, T., Eugster, W., Ojala, A., 2014. Eddy Covariance meaurement over lakes, in: Aubinet, M., Vesala, T., Papale, D. (Eds.), Eddy Covariance: A Practical Guide to Measurement and Data Analysis. Springer Atmospheric Sciences, pp. 164–168. https://doi.org/10.1002/9781118914236.ch13

Vick, S.H.W., Gong, S., Sestak, S., Vergara, T.J., Pinetown, K.L., Li, Z., Greenfield, P., Tetu, S.G., Midgley, D.J., Paulsen, I.T., 2019. Who eats what? Unravelling microbial conversion of coal to methane. FEMS Microbiol. Ecol. 95, 1–14. https://doi.org/10.1093/femsec/fiz093

Walter, K.M., Smith, L.C., Chapin, F.S., 2007. Methane bubbling from northern lakes: Present and future contributions to the global methane budget. Philos. Trans. R. Soc. A Math. Phys. Eng. Sci. 365, 1657–1676. https://doi.org/10.1098/rsta.2007.2036

Walter, P., Heimann, M., 2000. sites , sensitivity to model parameters , and climate + Qebull (t , + Qplant (t ,. New York 14, 745–765.

Wang, X., He, Y., Chen, H., Yuan, X., Peng, C., Yue, J., Zhang, Q., Zhou, L., 2018. CH4 concentrations and fluxes in a subtropical metropolitan river network: Watershed urbanization impacts and environmental controls. Sci. Total Environ. 622–623, 1079–1089. https://doi.org/10.1016/j.scitotenv.2017.12.054

Waschbusch, P., Korsch, R.J., Beaumont, C., 2009. Geodynamic modelling of aspects of the bowen, gunnedah, surat and eromanga basins from the perspective of convergent margin processes. Aust. J. Earth Sci. 56, 309–334. https://doi.org/10.1080/08120090802698661

Watson, A., Nedwell, D.B., 1998. Methane production and emission from peat. Atmos. Environ. 32, 3239–3245. https://doi.org/10.1016/s1352-2310(97)00501-3

Webb, J.R., Leavitt, P.R., Simpson, G, L., Baulch, H.M., Haig, H.A., Hodder, K.R., Finlay, K., 2019. Regulation of carbon dioxide and methane in small agricultural reservoirs: Optimizing potential for greenhouse gas uptake. Biogeosciences 16, 4211–4227. https://doi.org/10.5194/bg-16-4211-2019

Webster, M.A., Brew, J., Grimison, A.G., 2000. Hydrodynamics in the Queensland Sector of the Cooper/Eromanga Basins: Identifying Non-Conventional Exploration Plays Using Water Pressure and Chemistry Data. SPE - Asia Pacific Oil Gas Conf. 101–119. https://doi.org/10.2523/64281-ms

Wecht, K.J., Jacob, J.D., Frankenberg, C., Jiang, Z., Blake R, ., 2014. Journal of Geophysical Research : Atmospheres of SCIAMACHY satellite data. J. Geophys. Res. Atmos. Res. 7741–7756. https://doi.org/10.1002/2014JD021551.Received

West, William E., Coloso, J.J., Jones, S.E., 2012. Effects of algal and terrestrial carbon on methane production rates and methanogen community structure in a temperate lake sediment. Freshw. Biol. 57, 949–955. https://doi.org/10.1111/j.1365-2427.2012.02755.x

West, W. E., Coloso, J.J., Jones, S.E., 2012. Effects of algal and terrestrial carbon on methane production rates and methanogen community structure in a temperate lake sediment. Freshw. Biol. 57, 949–955. https://doi.org/10.1111/j.1365-2427.2012.02755.x

West, W.E., Creamer, K.P., Jones, S.E., 2016. Productivity and depth regulate lake contributions to atmospheric methane. Limnol. Oceanogr. 61, S51–S61. https://doi.org/10.1002/lno.10247

West, W.E., Mccarthy, S.M., Jones, S.E., 2015. Phytoplankton lipid content influences freshwater lake methanogenesis. Freshw. Biol. 60, 2261–2269. https://doi.org/10.1111/fwb.12652

Weyhenmeyer, C.E., 1999. Methane emissions from beaver ponds: Rates, patterns, and transport mechanism. Global Biogeochem. Cycles 13, 1079–1090.

Wik, M., 2016. Emission of methane from northern lakes and ponds.

Wik, M., Crill, P.M., Varner, R.K., Bastviken, D., 2013. Multiyear measurements of ebullitive methane flux from three subarctic lakes. J. Geophys. Res. Biogeosciences 118, 1307–1321. https://doi.org/10.1002/jgrg.20103

Wik, M., Thorton, B.F., Bastviken, D., MacIntyre, Varner, R.K., Crill, P.M., 2014. Bubbling in Subarctic Lakes. Geophys. Res. Lett. 555–560. https://doi.org/10.1002/2013GL058510.Received

Wilkinson, J., Maeck, A., Alshboul, Z., Lorke, A., 2015. Continuous Seasonal River Ebullition Measurements Linked to Sediment Methane Formation. Environ. Sci. Technol. 49, 13121–13129. https://doi.org/10.1021/acs.est.5b01525

Wu, Yanyou, 2021. Is bicarbonate directly used as substrate to participate in photosynthetic oxygen evolution. Acta Geochim. 40, 650–658. https://doi.org/10.1007/s11631-021-00484-0

Wu, Y., 2021. Bicarbonate use and carbon dioxide concentrating mechanisms in photosynthetic organisms. Acta Geochim. 40, 846–853. https://doi.org/10.1007/s11631-021-00488-w

Xiao, Q., Zhang, M., Hu, Z., Gao, Y., Hu, C., Liu, C., Liu, S., Zhang, Z., Zhao, J., Xiao, W., Lee, X., 2017. Spatial variations of methane emission in a large shallow eutrophic lake in subtropical climate. J. Geophys. Res. Biogeosciences 122, 1597–1614. https://doi.org/10.1002/2017JG003805

Xiao, S., Liu, D., Wang, Y., Yang, Z., Chen, W., 2013. Temporal variation of methane flux from Xiangxi Bay of the Three Gorges Reservoir. Sci. Rep. 3, 1–8. https://doi.org/10.1038/srep02500

Xiao, S., Yang, H., Liu, D., Zhang, C., Lei, D., Wang, Y., Peng, F., Li, Y., Wang, C., Li, X., Wu, G., Liu, L., 2014. Gas transfer velocities of methane and carbon dioxide in a subtropical shallow pond. Tellus, Ser. B Chem. Phys. Meteorol. 66, 1–14. https://doi.org/10.3402/tellusb.v66.23795

Xing, Y., Xie, P., Yang, H., Ni, L., Wang, Y., Rong, K., 2005. Methane and carbon dioxide fluxes from a shallow hypereutrophic subtropical Lake in China. Atmos. Environ. 39, 5532–5540. https://doi.org/10.1016/j.atmosenv.2005.06.010

Yago, J.V.., 1996. Basin Analysis of Middle Jurassic Walloon Coal Measures in the Great Artesian Basin, Australia. The University of Queensland.

Yang, L., Lu, F., Wang, X., Duan, X., Song, W., Sun, B., Zhang, Q., Zhou, Y., 2013. Spatial and seasonal variability of diffusive methane emissions from the Three Gorges Reservoir. J. Geophys. Res. Biogeosciences 118, 471–481. https://doi.org/10.1002/jgrg.20049

Young, A.L., 2005. Coalbed methane: A new source of energy and environmental challenges. Environ. Sci. Pollut. Res. 12, 318–321. https://doi.org/10.1065/espr2005.11.002

Yuan, J., Liu, D., Xiang, J., He, T., Kang, H., Ding, W., 2021. Methane and nitrous oxide have separated production zones and distinct emission pathways in freshwater aquaculture ponds. Water Res. 190, 116739. https://doi.org/10.1016/j.watres.2020.116739

Zhao, Y., Wu, B.F., Zeng, Y., 2013. Spatial and temporal patterns of greenhouse gas emissions from Three Gorges Reservoir of China. Biogeosciences 10, 1219–1230. https://doi.org/10.5194/bg-10-1219-2013

Zhilina, T.N., 1986. Methanogenic bacteria from hypersaline environments. Syst. Appl. Microbiol. 7, 216–222. https://doi.org/10.1016/S0723-2020(86)80009-1

Zielinska, B., Fujita, E., 1994. Organic gas sampling, in: Markert, B.A. (Ed.), Environmetnal Sampling for Trace Analysis. Wiley-Blackwell, Chichester, UK, pp. 163–184.

As Australia's national science agency and innovation catalyst, CSIRO is solving the greatest challenges through innovative science and technology.

CSIRO. Unlocking a better future for everyone.

Contact us

1300 363 400
+61 3 9545 2176
csiro.au/contact
csiro.au

For further information

Energy Business Unit
Se Gong
+61 2 9413 7885
se.gong@csiro.au

csiro.au/energy