General Comments:

This manuscript deals with an important issue, namely emissions of the strong greenhouse gas methane from the numerous small ponds in Queensland, Australia, in the context of climate changes and ponds number increase. Assessment of methane emissions supposes two steps: first the survey of small ponds and their relevant characteristics, second the assessment of methane emissions, depending on the pond characteristics (including size, type, location, climate and seasonality).

Reply: Thank you for highlighting the importance of this issue, we believe artificial ponds are a major gap in the development of greenhouse gas inventories and aimed to highlight their potential as a greenhouse gas source. We appreciate the review comments, particularly those associated with the need for a histogram, and the revised manuscript has greatly improved. Below are our detailed responses to the review comments. Page and Line numbers refer to the attached revised manuscript which contains track changes for changes suggested by both reviewers.

In this manuscript, the first step is realized based on existing data bases, the second on quite heavy emissions, and the dependence on emissions rate on inundated/non-inundated status of soil, the authors carried out complementary measures to characterize these causes of variability. The observations are then used to extrapolate emissions assessment to the global set of ponds, with two alternative extrapolation methods. This approach seems relevant. Anyway, several choices should have been discussed more in detail, and some underling hypothesis should have been clarified. In its present status, the manuscript is based on a large amount of data, not fully exploited. In particular, the way the "complementary" data is used (or not) is not clear.

Reply: Thank you for these comments. The issue of global ponds emissions will need to be addressed once further regional studies have been completed. Our primary objective with this study was to establish the greenhouse gas status of a broad range of ponds in one (large) region, and then to attempt to quantify the overall magnitude of emissions within this region. The 'complementary data' was used to understand the relative importance of diffusive and ebullition pathways and provide a measure of variability of our scaled emission estimates. This study did not seek to explore the drivers of emissions, this will form the basis of our future research in this area. Drivers of ebullition are difficult to elucidate even with detailed datasets given the non-linear interactions that cause bubble release from sediments. The manuscript has now been revised and we have clarified our objectives further on P3 L19-28:

"The principle objective of this study was to establish the GHG status of ponds in Queensland, Australia. Given the paucity of GHG data from ponds, this study has focussed on empirical assessments of CH_4 emissions from a range of pond types rather than detailed assessments of drivers of these emissions. Our assessment comprised four components:

1. Quantify the area of ponds, relative to regional assessments of larger artificial water bodies;
2. Quantify CH_4 emission rates for a wide spectrum of pond types;
3. Determine variability in their surface area and emission rates;
4. Determine the influence of inundation level on emission rates.

When integrated together, these components provide a robust regional assessment of anthropogenic CH_4 emissions for ponds in Queensland, Australia."

We have identified the relevant areas of future research on P11 L8-13:

"An additional consideration for future studies of ebullition patterns in ponds stems from recent studies of reservoirs which found significant changes in ebullition intensity and ebullition distribution as water levels decrease (Beaulieu et al., 2018;Hilgert et al., 2019). Under decreasing water levels, deeper zones of ponds may begin bubbling or increase the intensity of bubbling, this could potentially offset the reduction in surface available for emissions and total emissions would remain relatively constant."

And P12 L18-20:

"However, this finding was from a single urban lake and additional long term temporal studies along with high resolution spatial surveys of different pond types and size classes are required to identify the drivers of pond emission pathways."

Specific comments:

Here are some more specific comments illustrating the above comments:

Introduction:

Introduction could have also cited the other kinds of greenhouse gases y emitted by artificial ponds (CO2, N2O), and could have evoked their potential role of organic carbon sink. Balancing these two antagonist influences would give a larger context to the rest of the manuscript.

Reply: As suggested, we have now discussed CO_2 and N_2O emissions in the introduction. There are no studies of greenhouse gas emissions from artificial ponds in the region, however larger artificial water bodies that have been studied have demonstrated the dominance of methane in GHG emissions, hence our focus on this greenhouse gas. This has been included in the introduction on P1 L33-35:

"Whilst carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) can all be emitted, the most recent global synthesis of artificial water body emissions demonstrated that when converted to CO_2 equivalents, CH_4 accounted for 80% of fluxes (Deemer et al., 2016)."

And P2 L36-38:

"…(Downing, 2010;Deemer et al., 2016) but are in agreement with assessments of larger water bodies where CH_4 is the dominant GHG relative to N_2O and CO_2 (Merbach et al., 1996;Natchimuthu et al., 2014)."

The role of ponds as potential carbon sinks depends on the stability and permanence of organic carbon storage in sediments. In general they do not represent a stable long term sink, because they are routinely drained or the walls fail and the accumulated sediments are lost from the system. Furthermore, accumulations of organic sediment may enhance CH_4 emissions. These issues have now been discussed in more detail P2 L30-33:

"The potential of ponds as major organic carbon sinks has been established (Downing, 2010), although the stability and permanence of organic carbon trapped within ponds is critical to determining the magnitude of this sink. Loss pathways include active de-siltation (Verstraeten and Poesen, 2000), breaching of fully silted dams (Boardman and Foster, 2011) and methane emissions."

P2 - l40, it would be interesting to give some orders of magnitude of the CH4 sink effect of soils prior inundation, to compare with the emission rates presented later.

Reply: We have now included the order of magnitude rates for the methane sink in the manuscript, P3 L17:

"(ranging from -0.02 to -5 mg CH_4 m^{-2} d^{-1})"

P3-l5: it seems a 5th point is missing, which is the extrapolation of the four other points results in a regional emission assessment: at this is not straightforward, it should be mentioned. Point 3 is a bit misleading, as spatial and temporal variability in emission rate is in fact assessed only for one pond, independently of the pond's area variation.

Reply: We have removed the confusing terminology in component 3 and clarified that these four components are integrated into an overarching objective to quantify regional emissions from ponds in Queensland, Australia. This has section been revised on P3 L19-28:

"The principle objective of this study was to establish the GHG status of ponds in Queensland, Australia. Given the paucity of GHG data from ponds, this study has focussed on empirical assessments of CH_4 emissions from a range of pond types rather than detailed assessments of drivers of these emissions. Our assessment comprised four components:

1. Quantify the area of ponds, relative to regional assessments of larger artificial water bodies;
2. Quantify CH_4 emission rates for a wide spectrum of pond types;
3. Determine variability in their surface area and emission rates;
4. Determine the influence of inundation level on emission rates.

When integrated together, these components provide a robust regional assessment of anthropogenic CH_4 emissions for ponds in Queensland, Australia."

2.1 Study area description

The link between the fact that the majority of artificial water bodies are less than 5 ML and the choice to study emissions from ponds which area is less than 10^5 m² is not clear. Why this threshold?

Reply: Previous studies of farm dams have shown that 90% of stock watering dams are less than 5 ML but we have now removed this statement and focussed on the use of the 10^5 m^2 threshold. This has been revised on P4 L8-15:

"However, the number and surface area of ponds in Queensland is relatively unknown as there is no legal requirement to refer ponds to the state registry due to their small size. Under current state law only dam walls in excess of 10 m and volumes above 750 ML are referable (DEWS, 2017) and the maximum reported volume for ponds in Queensland is three times less than the referable volume (< 250 ML) (SKM, 2012). This study has assumed ponds are less than 100,000 m^2 as this is recognised globally as the major area of uncertainty in surface area assessments (Lehner and Doll, 2004;Downing 2010) and has been identified as a threshold in global lake inventories (Downing et al., 2006;Verpoorter et al., 2014)."

2.2 Relative surface area of ponds across the region

Given the discrepancy between the different sources of data and the difficulty to identify ponds and their characteristics within a large area, it would have been useful to give more details on the building of these three databases: how is data acquired, which kind of characteristic does each database include, at which periodicity is it revised, …

Reply: This illustrates a major challenge in developing regional greenhouse gas inventories that incorporate ponds. The two State Government databases (Reservoirs and Water Storage Points) were developed for Queensland to better understand the hydrology of the state in response to the Millennium Drought (1997 to 2009). They included a range of different aerial imagery (10 to 60 cm orthophotography) and satellite products (0.5 to 2.5 m resolution) in a major effort over the period of 2010 to 2014. Despite the availability of these high resolution databases these have still not been fully integrated into the official land use assessment for Queensland and represent a major area of uncertainty in the surface area available for emissions. This is the first time such an exercise has been attempted so the periodicity at which it will be revised is difficult to estimate. However, with high resolution satellite imagery becoming increasingly available this will allow the total surface area of ponds to be revised more frequently.

We have added additional information on these databases and how they were used in P4 L21-23:

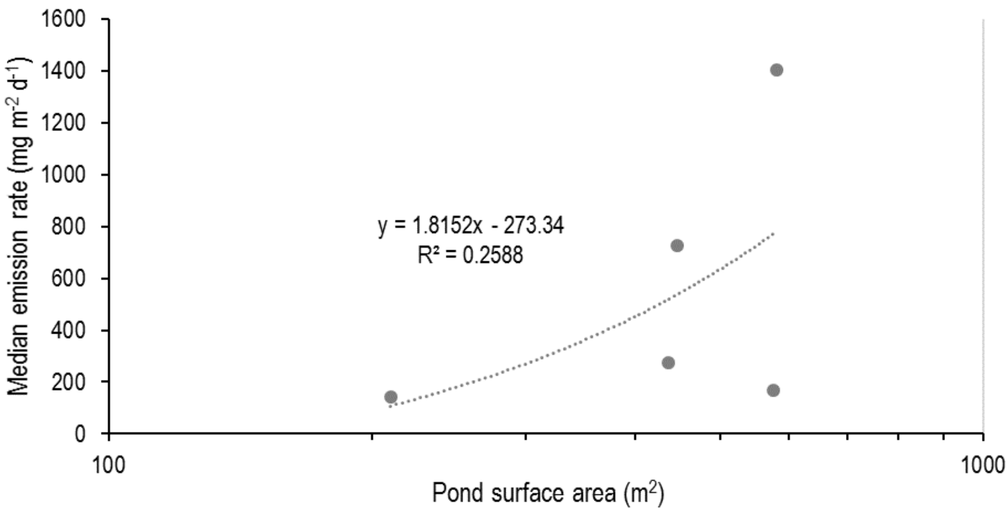
"To determine the number and relative surface area of ponds across the Queensland, three State Government GIS databases of artificial water bodies were utilised. However these databases required additional processing to extract comparable pond data as there were inconsistencies in the format and nomenclature of feature types."

And P4 L31-32:

"Both databases are derived from aerial (10 to 60 cm orthophotography) and satellite (0.5 to 2.5 m resolution) imagery captured between 2010 and 2014."

Given the strong dependency of CH4 emission rate on the ponds' size, why choosing an average area for ponds <625m², instead of using a size distribution. The same question arises for the classification in the three classes of the GRanD. This seems premature before the later results. Anyhow, it would have been interesting to present a histogram of the ponds' sizes. It is also disappointing not to know more about the types of ponds (and the potential link between type and size), their location, the way they are supplied in water, … all characteristic which may influence methane emissions and which may be available in the databases?

Reply: We share the frustration at not having more information about the type of ponds, but for both databases there was very limited classification of feature types: Reservoirs were divided into feature types: Flood Irrigation Storage, Rural Water Storage and Town Water Storage. Water Storage Points only had one feature type for all points: Dam. The ponds in Water Storage Points database were classified in the 10^2 to 10^3 m^2 size class and there was a very weak relationship between median emission rate and pond surface area (see figure below). For this reason we preferred to use a mean surface area for these systems.



We have also included a histogram showing the relationship between the three major pond types and their sizes in the Appendix P30 Figure A2:

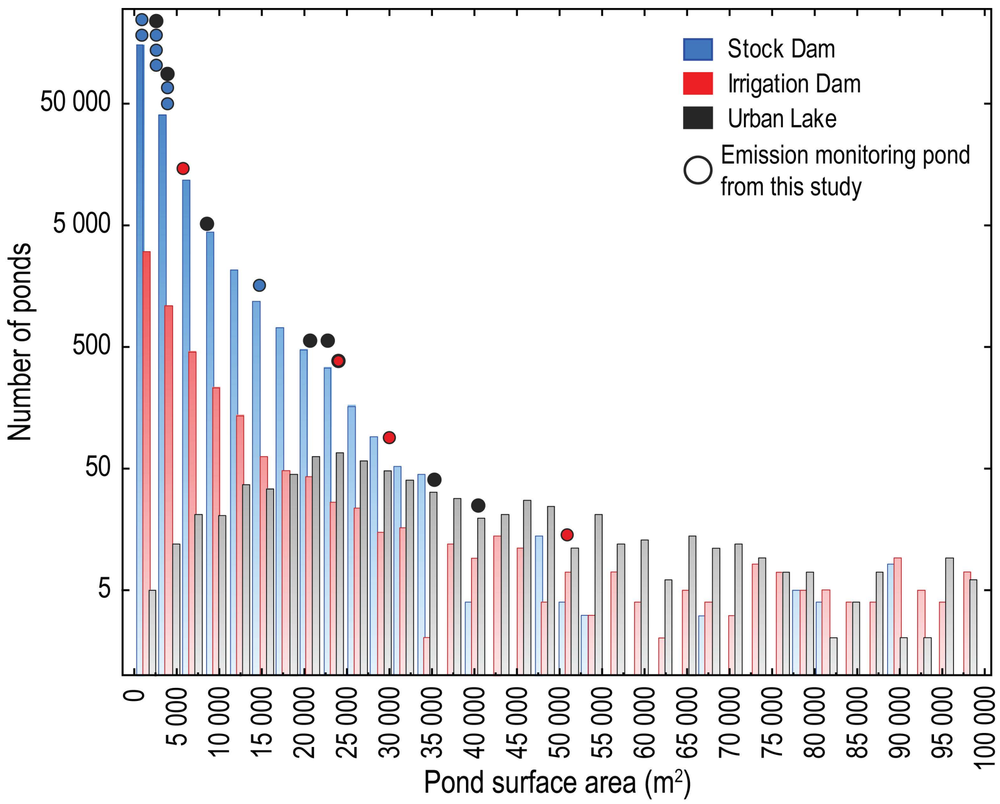


Figure A2. Pond size from emission study relative to histogram of regional pond distribution of stock dams, irrigation dams and urban lakes. The surface area of pond used emission study (Table A1). Histogram of regional distribution of ponds was developed from QLUMP, Reservoir and Water Storage Points databases and separated into pond type depending on surrounding land use: "Grazing native vegetation" for stock dams; "Production from irrigated agriculture and plantations" for irrigation dams; "Intensive uses" for urban lakes with "Mining" and "Manufacturing" landuse within "Intensive Uses" were removed to ensure only urban areas were selected. To incorporate the distribution of ponds within the Water Storage Points database, it

was assumed this would match the distribution from the 100 individual ponds examined in Section 2.2 to determine their average surface area.

2.3 CH4 emissions from broad spectrum of pond types

It would have been useful for readers not familiar with methane emission from waterbodies to present the different kind of methane emissions measures, their advantages and limits, and to argument the choice performed here. Above all, the choice of the studied ponds should have been discussed, and their representativeness of the whole ponds set variety assessed.

Reply: We have now included this suggestion and revised the methodology to include the relative advantages and disadvantages of commonly used methods to assess methane emissions and to justify the approach we took in our choice to use floating chambers. These changes are on P5 L23-25:

"Stock dams, irrigation dams and urban lakes account for the vast majority of ponds across Queensland and ponds within each category were selected to represent the regional size class distribution (Fig. A2)."

And P5 L27-33:

"There are a number of commonly used methods to assess methane emissions from water bodies depending on the pathway of interest. For the diffusive emission pathway, rates may be modelled using the thin boundary methods or directly measured using manual or automatic floating chambers (St. Louis et al., 2000). For ebullition pathways, rates can be directly measured using acoustic surveys or funnel traps (DelSontro et al., 2011). Thin boundary layer models cannot be used to quantify the ebullition pathway and acoustic surveys or funnel traps cannot be used effectively in ponds as the water depth is often too shallow (< 1 m). We chose to use floating chambers to capture both ebullition and diffusive fluxes."

Small, light chambers had an additional advantage in their relative ease of deployment in these relatively challenging environments, examples of which are now provided in the Appendix P31 Figure A4:

Figure A4. Oblique drone images showing natural obstacles for pond chamber deployments from a) emergent macrophytes and b) floating aquatic weeds.

The representativeness of the studied ponds relative to the whole region is now shown on the histogram in the Appendix P30 FigureA2, where the studied ponds captured the regional surface area distribution for major pond types.

How was the number of floating chambers per pond chosen? It seems not to be only in function on the pond's size? Was the uncertainty arising from measuring only 6 to 8 hours of emission for 3 ponds assessed? Is the emission process known to be varying at the daily scale? When did the monitoring occurred during the year (and which year rather dry or wet)?

Reply: We have now included more detail in the methodology to address these comments. The number of chambers deployed per pond was a function of both the pond size and local restrictions on access. For each pond a minimum of 3 chambers were deployed along a transect to ensure deep and shallow areas were monitored. The uncertainty arising from 6 to 8 hours deployment was not assessed, however, studies to date have shown diffusive emissions may be subject to diurnal changes but ebullition (typically the largest emission pathway) does not undergo regular diurnal change, so there should not be a significant bias. Of the three ponds where 6 to 8 hour incubation were undertaken, the diffusive pathway was only dominant at Lake Alford. More detail on the uncertainty has been stated in P6 L2-8:

"The 24 hour deployment time was chosen to increase the likelihood to capture ebullition, which is episodic in nature, and to incorporate diel variability in diffusive emissions which can be up to a 2-fold bias (Bastviken et al., 2004;Bastviken et al., 2010;Natchimuthu et al., 2014). The use of long term deployments may underestimate diffusive fluxes, which decrease as the chamber headspace approaches equilibrium with the water. However, in contrast to CO_2, CH_4 has a long equilibration time and it has been shown that a 24 hour deployment of these types of flux chambers on lakes underestimate diffusive fluxes by less than 10% (Bastviken et al., 2010)."

The annual study was undertaken in 2017 and the broad water body survey measurements were undertaken in August to December 2017. This has been stated P5 L20-21 ("from August to December 2017"). Rainfall across Queensland in 2017 was 10% below average.

Nothing is said about the way the pond global emission rate is assessed from the punctual measures: given the variability at the pond scale illustrated with 2.4.1 results, it yet seems crucial. Uncertainty arising from how this calculation was performed may be as high as those arising from the choice of arithmetical or geometrical means between ponds' emission rates later on (2.6).

Reply: We took the approach to use the average surface area at full supply level (A_{FSL}) for respective size classes. We believe the highest uncertainty lies with the emissions rates as these varied by over an order of magnitude whereas as the surface area variability, particularly in the larger pond size classes was relatively stable, and the mean ranged only 5%. The uncertainty associated with surface area changes is buffered by the consistency of emissions from deeper zones which experience less frequent drying periods. These comments are now included in the discussion sections P11 L8-13:

"An additional consideration for future studies of ebullition patterns in ponds stems from recent studies of reservoirs which found significant changes in ebullition intensity and ebullition distribution as water levels decrease (Beaulieu et al., 2018;Hilgert et al., 2019). Under decreasing water levels, deeper zones of ponds may begin bubbling or increase the intensity of bubbling, this could potentially offset the reduction in surface available for emissions and total emissions would remain relatively constant."

2.4 Spatial and temporal variability in surface area and emission rate

2.4.1

Here also it is not clear why this pond was chosen rather than another. What about its representativeness? For example, one can expect that the temporal variability of a weir's emission rate to be higher than an urban's lake one? The year when this monitoring was performed is not specified, neither corresponding rainfall, which may be of influence on the pond supply and the emission processes? Air and water temperature may also be influent factors?

Reply: This pond was chosen for both its ease of access as well as relatively stable surface area allowing long term monitoring of the same site. This has been clarified on P6 L17-19:

"This pond was selected as water level remains relatively constant throughout the year and sampling would not impacted by changes in inundation status."

It provides a typical example of an urban lake but we recognise and acknowledge that we would need measurements from examples of each pond type to provide a truly representative dataset. Rainfall at the urban lake site was average for 2017 and there were no major maintenance programs undertaken on the lake during this time. Our aim was to obtain indicative and upscaleable estimates of emissions in this study, not to fully elucidate the mechanisms determining temporal variability which would require a different approach. This will form the basis of future studies for our research in this area. We have now adjusted the future research section to include the need for focussed studies to establish the major drivers of emissions within each pond type P12 L18-20:

“However, this finding was from a single urban lake and additional long term temporal studies along with high resolution spatial surveys of different pond types and size classes are required to identify the drivers of pond emission pathways.”

2.4.2

Is rainfall variability homogeneous at the state scale? In other words, are the percentages of AFSL calculated for the ponds which were monitored relevant for the regional scaling which is performed in 4.1?

Reply: Rainfall is not homogenous at the state scale, with clear zones of variability following the isohyets shown in Figure 2 b. Low rainfall areas (< 300 mm yr^{-1}) particularly in the western parts have the highest variability, whilst zones (> 300 mm yr^{-1}) closer to the coastal regions have low to moderate variability (http://www.bom.gov.au/jsp/ncc/climate_averages/rainfall-variability/index.jsp). Over 90% of the state’s ponds are located within the bands of low to moderate rainfall variability and would support regional scaling from the ponds measured in this study. To calculate the percentage of A_{FSL} imagery was collected spanning the time period 2009 to 2017 which includes the both drought and flood years (P6 L35-37) and would cover the expected range in rainfall across the state and provides additional confidence in the regional scaling.

2.5

As for the 2.4.1 section, it is not clear why this pond was chosen rather than another. What assures its representativeness of other small ponds which area varies a lot depending on rainfall?

Reply: This pond was selected to represent stock dams which are the most numerous pond type and represent the largest surface area (Appendix P30 Figure A2). The construction of thi pond is typical of those in the region (a shallow pit is dug out and the soil used to construct the wall and spillway) and the surface area closely matched the median for all farm dams. The variability in surface area for stock ponds in this region is primarily dependent on whether they are being used stock watering, rather than rainfall. The site justification is now provided on P7 L5-9:

“This pond was selected as stock dams generally experience accelerated rates of water level change due to their relatively small size compared to other pond types (Fig. A2). In addition, the construction of this pond is typical for stock dams (a shallow pit is dug out and the soil used to construct the wall and spillway) and the surface area (1,893 m^2) closely matched the median for all farm dams (1,586 m^2; Fig. A2).”

These complementary field campaigns are honest and interesting attempts to deepen the study, but should be more detailed and argued to be totally useful and convincing to strengthen the results which are the main scope of the paper.

Reply: We are very grateful for your insights and have clarified the introduction and study objectives to support our approach. This can be found on P2 L8-11:

“In addition, the peripheral areas of small water bodies regularly experience periods of inundation and no inundation as water levels change due to their relatively shallow nature and high water use rates. The changes in their inundation status may influence emission rates as has been observed for natural ponds (Boon et al 1997).”

And P3 L4-6:

“An important part of the value of building a dataset of CH_4 flux estimates from a broad range of sites is determining factors that account for spatial and temporal variability in the flux.”

And P3 L19-21:

“The principle objective of this study was to establish the GHG status of ponds in Queensland, Australia. Given the paucity of GHG data from ponds, this study has focussed on empirical assessments of CH_4 emissions from a range of pond types rather than detailed assessments of drivers of these emissions.”

Furthermore, we revised the methods, results and future studies to demonstrate the utility of these complementary studies. This is stated in the following sections on:

P6 L14-16:

“To gain insight into the spatial and temporal uncertainty in pond emissions we compared variability in seasonal emissions from a single site to emissions from an intensive spatial survey of multiple sites across the pond (Fig. 4).”

P9 L31-33:

“In contrast emissions from central areas were over 100 mg m^{-2} d^{-1}, more than double the peripheral area emission rates (Table A1).”

P12 L14-15:

“However, this was limited to a single stock dam and additional pond types and size classes must be examined before more confident generalisations can be made.”

P12 L18-20:

“However, this finding was from a single urban lake and additional long term temporal studies along with high resolution spatial surveys of different pond types and size classes are required to identify the drivers of pond emission pathways.”

3.2 CH4 emissions from ponds

Some considerations on ebullition/diffusive emissions would be necessary to support the conclusion that ebullition is the dominant emission pathway. If so, spatial heterogeneity of emission must be high: is the monitoring protocol adapted to capture this heterogeneity?

Reply: We agree that it is important to capture both the spatial and temporal heterogeneity in the ebullition pathway. We believe that our approach to follow a transect across deep and shallow areas should capture this spatial variability at a broad level, and that the 24 hour deployments would maximise the likelihood of capturing episodic ebullition events. We accept that both the total CH_4 emissions and the balance of diffusion and ebullition are spatially variable and uncertain, and have acknowledged this in the manuscript P5 L27-33:

“There are a number of commonly used methods to assess methane emissions from water bodies depending on the pathway of interest. For the diffusive emission pathway, rates may be modelled using the thin boundary methods or directly measured using manual or automatic floating chambers (St. Louis et al., 2000). For ebullition pathways, rates can be directly measured using acoustic surveys or funnel traps (DelSontro et al., 2011). Thin boundary layer models cannot be used to quantify the ebullition pathway and acoustic surveys or funnel traps cannot be used effectively in ponds as the water depth is often too shallow (< 1 m). We chose to use floating chambers to capture both ebullition and diffusive fluxes."

And P5 L37-39:

"The floating chambers used were designed to yield negligible bias on the gas exchange and compare well with non-invasive approaches (Cole et al., 2010;Gålfalk et al., 2013;Lorke et al., 2015)."

And P6 L2-8:

"The 24 hour deployment time was chosen to increase the likelihood to capture ebullition, which is episodic in nature, and to incorporate diel variability in diffusive emissions which can be up to a 2-fold bias (Bastviken et al., 2004;Bastviken et al., 2010;Natchimuthu et al., 2014). The use of long term deployments may underestimate diffusive fluxes, which decrease as the chamber headspace approaches equilibrium with the water. However, in contrast to CO_2, CH_4 has a long equilibration time and it has been shown that a 24 hour deployment of these types of flux chambers on lakes underestimate diffusive fluxes by less than 10% (Bastviken et al., 2010)."

On the other hand, existing data on CH_4 emissions from artificial ponds are extremely scarce, to the extent that this study alone will considerably increase the total global dataset of measurements. We hope that it will therefore represent an important step towards developing a more comprehensive understanding of the role ponds play in greenhouse gas emissions and the carbon cycle. This has been included in sections introduction and future areas of research:

P2 L29-33:

"In addition, quantifying methane emission from ponds will improve our understanding of their role in the global carbon cycle. The potential of ponds as major organic carbon sinks has been established (Downing, 2010), although the stability and permanence of organic carbon trapped within ponds is critical to determining the magnitude of this sink. Loss pathways include active de-siltation (Verstraeten and Poesen, 2000), breaching of fully silted dams (Boardman and Foster, 2011) and methane emissions."

And P11 L34-35:

"… will greatly improve the surface area estimate of flooded lands used for upscaling greenhouse gas emissions as well as their role in the global carbon cycle."

As said before, a histogram of sizes and types of ponds would be useful to contextualize the results. Detail of the way the emissions are assessed at the pond scale too.

Reply: This is an excellent suggestion, thank you - we have now added the suggested histogram, which indicates that our sample sites closely reflect the mix of pond types and areas within our study region Appendix P30 Figure A2.

As weirs present high emission rates compared to other types of ponds, is it relevant to group them with other small ponds/stock ponds to assess the regional scale emissions?

Reply: We accept this comment, and have now excluded weirs from the regional emissions scaling and revised table accordingly (P18 Table 1). The regional contribution decreased by less than 1% and the study conclusions remain valid. Revised text is found on P18 L8 "however, weir emissions were omitted as these are not relevant at the regional scale."

3.3.1 Spatial and temporal variability within a single pond

These data are no doubt very interesting, but what they bring to this study is not clear for me. How are they used to the following regional scaling? If it is by considering that observed emissions from the other ponds can be taken as annual averages, this should be clarified. If this is the case, the validity of this hypothesis should be discussed, as only one type of pond was considered for this analysis. It seems to me that this part could be cut from this manuscript, and maybe give the material for another paper, which would allow to analyse more in depth the emissions variability and the factor which influence it.

Reply: We have now clarified the text to highlight the limitations of extrapolating these results across different pond types. The value we see in these data is demonstrating ebullition across the annual cycle at a single point, suggesting this is a persistent feature when conditions favour it. Additional text can be found on P12 L18-20:

"However, this finding was from a single urban lake and additional long term temporal studies along with high resolution spatial surveys of different pond types and size classes are required to identify the drivers of pond emission pathways."

The spatial and temporal patterns observed in these data do provide striking similarities with larger reservoirs in the region supporting the persistence of the ebullition pathway across the annual cycle and the importance of spatial variability in emissions rates. Both of which we consider valuable insights into the emissions from a smaller water body. On balance we would therefore prefer to retain this part of the manuscript.

4.1

In India, numerous ponds are used to increase groundwater discharges. As a consequence, CH4 emission from these ponds may differ from the types of ponds which were studied here. This could be specified to be rigorous in this discussion about the importance to take into account ponds in methane emission rates assessment.

Reply: Thank you for this suggestion and we have now included a section to highlight the importance of local pond types to regional emissions P12 L4-7:

"An additional consideration is to ensure pond emission studies from different regions include all relevant ponds types. For example, the use of ponds to increase groundwater recharge is widespread across South East Asia (Giordano, 2009) and these would need to be included in regional inventories."

4.2

Again, more details should have been given above on the different emission pathways and their influencing factors.

Reply: We have now included sections providing more details on the major pathways and their influencing factors as well as key areas for future research to address these issues. These sections can be found on P11 L8-13:

"An additional consideration for future studies of ebullition patterns in ponds stems from recent studies of reservoirs which found significant changes in ebullition intensity and ebullition distribution as water levels decrease (Beaulieu et al., 2018;Hilgert et al., 2019). Under decreasing water levels, deeper zones of ponds may begin bubbling or increase the intensity of bubbling, this could potentially offset the reduction in surface available for emissions and total emissions would remain relatively constant."

And P12 L4-7:

"An additional consideration is to ensure pond emission studies from different regions include all relevant ponds types. For example, the use of ponds to increase groundwater recharge is widespread across South East Asia (Giordano, 2009) and these would need to be included in regional inventories."

And P12 L18-20:

"However, this finding was from a single urban lake and additional long term temporal studies along with high resolution spatial surveys of different pond types and size classes are required to identify the drivers of pond emission pathways."

As depth, the way water is supplied in pond and substrate seem to be influential, this discussion could address the question of the availability of such data in current databases.

Reply: Thank you for this comment and, unfortunately, no depth data or the water delivery information is provided in the databases. Depth can be inferred from surface area to volume relationships such as that reported in Lowe et al 2005. Water supply occurs through a number of different pathways including overland flow from rainfall, direct pumping from river and stormwater inflow from artificial drainage networks.

Lowe, L., Nathan, R., and Morden, R. 2005. Assessing the impact of farm dams on streamflows, Part II: Regional characterisation, Australasian Journal of Water Resources, 9, 13-26.

5 Future research

Some sources of uncertainties are taken into account in this manuscript, other are not: this section could emphasize these points, and discuss how to handle them (assessment of a whole pond emission rate given punctual data in time and space; identification of emission pathways, characterisation of the way some factors influence emission rates–type, depth, purpose, water supply, temperature …) . At the moment, the research perspectives are more or less an extension of the work which was already performed.

Reply: We have now included a broader discussion of the sources of uncertainty, as suggested, as well as the priority research needs required to reduce this uncertainty and improve our understanding of pond's contribution to flooded land emissions and loss pathways of sequestered carbon. New text has been added in the following sections:

P2 L29-33:

"In addition, quantifying methane emission from ponds will improve our understanding of their role in the global carbon cycle. The potential of ponds as major organic carbon sinks has been established (Downing, 2010), although the stability and permanence of organic carbon trapped within ponds is critical to determining the magnitude of this sink. Loss pathways include active de-siltation (Verstraeten and Poesen, 2000), breaching of fully silted dams (Boardman and Foster, 2011) and methane emissions."

P11 L8-13:

"An additional consideration for future studies of ebullition patterns in ponds stems from recent studies of reservoirs which found significant changes in ebullition intensity and ebullition distribution as water levels decrease (Beaulieu et al., 2018;Hilgert et al., 2019). Under decreasing water levels, deeper zones of ponds may begin bubbling or increase the intensity of bubbling, this could potentially offset the reduction in surface available for emissions and total emissions would remain relatively constant."

And P12 L4-7:

"An additional consideration is to ensure pond emission studies from different regions include all relevant ponds types. For example, the use of ponds to increase groundwater recharge is widespread across South East Asia (Giordano, 2009) and these would need to be included in regional inventories."

And P12 L14-15:

"However, this was limited to a single stock dam and additional pond types and size classes must be examined before more confident generalisations can be made."

And P12 L18-20:

"However, this finding was from a single urban lake and additional long term temporal studies along with high resolution spatial surveys of different pond types and size classes are required to identify the drivers of pond emission pathways."

Incorporating the comments from both reviews has improved this manuscript and highlighted the importance of this work in highlighting the role ponds likely play in global greenhouse gas emissions.

The importance of small artificial water bodies as sources of methane emissions in Queensland, Australia.

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Abstract. Emissions from flooded land represent a direct source of anthropogenic greenhouse gas emissions. Methane emissions from large, artificial water bodies have previously been considered, with numerous studies assessing emission rates and relatively simple procedures available to determine their surface area and generate upscaled emissions estimates. In contrast, the role of small artificial water bodies (ponds) is very poorly quantified, and estimation of emissions is constrained both by a lack of data on their spatial extent, and a scarcity of direct flux measurements. In this study, we quantified the total surface area of water bodies $<10^5$ m^2 across Queensland, Australia, and emission rates from a variety of water body types and size classes. We found that the omission of small ponds from current official land-use data has led to an under-estimate of total flooded land area by 24%, of small artificial water body surface area by 57%, and of the total number of artificial water bodies by an order of magnitude. All studied ponds were significant hotspots of methane production, dominated by ebullition (bubble) emissions. Two scaling approaches were developed with one based on pond primary use (stock watering, irrigation and urban lakes) and the other using size class. Both approaches indicated that ponds in Queensland alone emit over 1.6 Mt CO_2-eq yr^{-1}, equivalent to 10% of the state's entire Land Use, Land Use Change and Forestry sector emissions. With limited data from other regions suggesting similarly large numbers of ponds, high emissions per unit area, and under-reporting of spatial extent, we conclude that small artificial water bodies may be a globally important 'missing source' of anthropogenic greenhouse gas emissions.

1 Introduction

Over the last 20 years greenhouse gas emissions studies from large, artificial water bodies such as water supplies or hydroelectric reservoirs have clearly demonstrated these are major emissions sources~~ of atmospheric methane (CH_4) emissions~~. Whilst carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) can all be emitted, the most recent global synthesis of artificial water body emissions demonstrated that when converted to CO_2 equivalents, CH_4 accounted for 80% of fluxes (Deemer et al., 2016). Increasingly sophisticated reviews have explored the magnitude of the artificial water body~~their~~ contribution to regional and global CH_4 budgets (St. Louis et al., 2000;Bastviken et al., 2011;Deemer et al., 2016). Much of the focus in reducing the uncertainty from this anthropogenic greenhouse gas source has focussed on the spatial and temporal variability in total emission rates and, in particular, the relative contribution of CH_4 bubbling (ebullition) directly from the sediment (Bastviken et al., 2011). To enable large-scale emissions estimates from larger, artificial waterbodies, relationships between eutrophication status and sediment temperature (Aben et al., 2017;Harrison et al., 2017) have been developed to predict both diffusive and ebullitive emission rates. However, in regional or global scaling of emissions it is important to examine the emission rates of all types and sizes of artificial water bodies (Panneer Selvam et al., 2014). Furthermore the surface area of small water bodies can be particularly difficult to quantify in national and global datasets due to their small size and large number~~it is important to examine the uncertainty in surface area~~ (Chumchal et al., 2016). In addition, the peripheral areas of small water bodies regularly experience periods of inundation and no inundation as water levels change due to their relatively shallow nature and high water use rates. The changes in their inundation status may influence emission rates as has been observed for natural ponds (Boon et al 1997). ~~and the emission rates of all classes of artificial water bodies (Panneer Selvam et al., 2014).~~ Given there are estimated to be 16 million artificial water bodies with a surface area less than 0.1 km^2 (Lehner et al., 2011), understanding the rates and variability in emissions from these flooded lands will be an important refinement to global CH_4 budgets.

The increasing urbanisation of society as well as the expansion of agriculture and commercial mining activities has resulted in a proliferation of small artificial water bodies in many parts of the globe (Renwick et al., 2005;Downing et al., 2006;Pekel et al., 2016). This is well illustrated by the example from the United States where artificial small water bodies increased from an estimated 20,000 in 1934 (Swingle, 1970) to over 9 million in 2005 (Renwick et al., 2005). These water bodies provide valuable services and are required to irrigate crops, provide water for farm stock, manage stormwater, offer visual amenity and recreational activities, and supply water for industrial processes (Fairchild et al., 2013). Small water bodies are often avian biodiversity hotspots, for example hosting an estimated 12 million water birds in a single catchment area in the Murray-Darling river system, Australia (Hamilton et al., 2017).

The creation ~~However, the creation~~ of ~~water~~ small artificial water bodies also represents a transformation of the landscape, referred to in the Intergovernmental Panel on Climate Change land-use emission accounting procedures as 'Flooded Lands' (IPCC, 2006). Where the creation of small water bodies leads to new greenhouse gas (GHG) emissions, these emissions are considered anthropogenic in origin according to IPCC guidelines (IPCC, 2006), ~~these can be considered anthropogenic in origin~~, and should therefore be included in Flooded Lands emissions inventories (Panneer Selvam et al., 2014). In addition, quantifying methane emission from ponds will improve our understanding of their role in the global carbon cycle. The potential of ponds as major organic carbon sinks has been established (Downing, 2010), although the stability and permanence of organic carbon trapped within ponds is critical to determining the magnitude of this sink. Loss pathways include active de-siltation (Verstraeten and Poesen, 2000), breaching of fully silted dams (Boardman and Foster, 2011) and methane emissions.

To date, the relatively few regional studies on small, artificial water bodies (hereafter 'ponds') have focussed on water and sediment dynamics rather than GHG emissions (Downing et al., 2008;Callow and Smettem, 2009;Verstraeten and Prosser, 2008;Habets et al., 2014). Studies of ~~CH_4 or other~~ GHG emissions from ponds have been limited (Downing, 2010;Deemer et al., 2016) but~~,~~ are in agreement with assessments of larger water bodies where CH_4 is the dominant GHG relative to N_2O and CO_2 (Merbach et al., 1996;Natchimuthu et al., 2014).~~and many are restricted to fairly short-term measurements at a small number of sites within a limited geographical area (Downing, 2010;Deemer et al., 2016).~~ The only regional-scale study to date was undertaken in India by

Panneer Selvam et al. (2014). In order to quantify the role of artificial ponds in the global CH_4 cycle, as well as their role as a source of anthropogenic emissions, it is necessary ~~both~~ to obtain both estimates of CH_4 fluxes from a broader range of sites ~~(and to determine the factors that account for spatial and temporal variability in flux)~~ and also to estimate the surface area contributing to emissions. An important part of the value of building a dataset of CH_4 flux estimates from a broad range of sites is determining factors that account for spatial and temporal variability in the flux. Surface area estimates can be problematic given the range of water types (small urban lakes to large irrigation ponds) that fall within the definition of 'ponds', their frequently high temporal variation in surface area, the sheer number of such water bodies, and their ongoing increase in number over time.

Here, we present the first regional-scale assessment of CH_4 emissions from ponds in the Southern Hemisphere and, following the assessment of Panneer Selvam et al., (2014) only the second regional assessment globally. The assessment was undertaken in the 1.85 million km^2 State of Queensland, Australia. Queensland provides an effective test case for the estimation of CH_4 emissions from ponds because i) it incorporates a high degree of spatial variability in land-use and climate, from desert to humid tropics; and ii) the irregular rainfall patterns and wide spatial coverage of aerial imagery result in a large number of artificial ponds, which are relatively easy to quantify. CH_4 emissions from these ponds can be considered anthropogenic in origin, because past studies of rainforest and agricultural soils in the region have clearly shown these terrestrial landscapes were weak CH_4 sinks (ranging from -0.02 to -5 mg CH_4 m^{-2} d^{-1}) prior to inundation (Allen et al., 2009;Scheer et al., 2011;Rowlings et al., 2012).

The principle objective of this study was to establish the GHG status of ponds in Queensland, Australia. Given the paucity of GHG data from ponds, this study has focussed on empirical assessments of CH_4 emissions from a range of pond types rather than detailed assessments of drivers of these emissions. Our assessment comprised four components:

1. Quantify the area of ponds, relative to regional assessments of larger artificial water bodies;
2. Quantify CH_4 emission rates for a wide spectrum of pond types;
3. Determine variability in their surface area and emission rates;
4. Determine the influence of inundation level on emission rates.

When integrated together, these components provide a robust regional assessment of anthropogenic CH_4 emissions for ponds in Queensland, Australia.

~~Our assessment comprised four components, designed to quantify the total anthropogenic CH_4 emission from ponds in Queensland, as well as their variability:~~

1. ~~Quantify the area of ponds, relative to regional assessments of larger artificial water bodies;~~
2. ~~Quantify CH_4 emission rates for a wide spectrum of pond types;~~
3. ~~Determine spatial and temporal variability in their surface area and emission rates;~~
4. ~~Determine the influence of inundation level on emission rates.~~

2 Methodology

2.1 Study area description

Queensland, the second largest state in Australia, covers a surface area of 1.85 million km^2 and ~~having~~ has a population of 4.75 million people. Land use across the state is dominated by agriculture with over 80% of the total surface area utilised for grazing cattle or irrigated cropping (Fig. 2 a; QLUMP, 2018). The Queensland agriculture sector contributes more than AUD$13 billion per year to the state economy and includes 15 million cattle and sheep as well as 4,526 km^2 of land under irrigation (ABS, 2018). The climate is subtropical or tropical with mean annual temperatures ranging from 27.5 ºC in the state's north to 15.8 ºC in the southern interior. There are large gradients in rainfall across the state ranging from a mean annual rainfall of over 3,000 mm in the coastal north east to less than 100 mm in the arid western regions (Fig. 2 b). Rainfall has a distinct annual pattern with up to 80% falling during the summer months from November to April and is subject to ~~major~~ decadal drought and flood cycles ~~at decadal cycles~~ (Klingaman et al., 2013). The economic importance of agriculture coupled with the need to provide year round water supply for these activities and the lack of predictable rainfall has resulted in the proliferation of artificial water bodies across the ~~whole~~ state (Fig. A1). However, the number and surface area of ponds in Queensland is relatively unknown as there is no legal requirement to refer ponds to the state registry due to their small size. Under current state law only dam walls in excess of 10 m and volumes above 750 ML are referable (DEWS, 2017) and the maximum reported volume for ponds in Queensland is three times less than the referable volume (< 250 ML) (SKM, 2012). This study has assumed ponds are less than 100,000 m^2 as this is recognised globally as the major area of uncertainty in surface area assessments (Lehner and Doll, 2004;Downing 2010) and has been identified as a threshold in global lake inventories (Downing et al., 2006;Verpoorter et al., 2014). ~~Under current state law only dam walls in excess of 10 m and 750 ML are referable to the state registry (DEWS, 2017) with 109 dams registered (Referable dams register; http://qldspatial.information.qld.gov.au/catalogue/). The vast majority of artificial water bodies are less than 5 ML (Nathan and Lowe, 2012) and these ponds, therefore, represent an area of major uncertainty in the assessment of land use change assessment and associated greenhouse gas emissions.~~

2.2 Relative surface area of ponds across the region

To determine the number and relative surface area of ponds across the Queensland, three State Government GIS databases of artificial water bodies were utilised. However these databases required additional processing to extract comparable pond data as there were inconsistencies in the format and nomenclature of feature types. The ~~number and relative surface area of ponds in Queensland was derived from~~ primary database used was the most recent official assessment of land use from March 2018 (QLUMP, 2018) and~~. W~~ within the Primary land use classification of "Water" there is a secondary category of artificial "Reservoirs/dam" divided further into "Reservoirs, Water storage and Evaporation basin." The individual water body surface area is provided and all ~~reservoirs and water storages~~ponds ~~less than 100,000~~ (≤10^5~~)~~ m^2) were extracted from the database. Evaporation basins were excluded, as these are commonly used for salt extraction. These ponds were then compared against two State Government databases from a high resolution assessment of artificial water bodies across the state published in 2014 and 2015. Both databases are derived from aerial (10 to 60 cm orthophotography) and satellite (0.5 to 2.5 m resolution) imagery captured between 2010 and 2014. One database contains water bodies greater than 625 m^2 at full supply (Reservoirs – Queensland; http://qldspatial.information.qld.gov.au/catalogue/) and for water bodies less than 625 m^2 a second database was used (Water storage points - Queensland; http://qldspatial.information.qld.gov.au/catalogue/).

Water bodies larger than 625 m^2 contained individual polygons where water body surface area was provided and all water bodies less than 10^5 m^2 were extracted from the database. The database of water bodies smaller than 625 m^2 contained point data providing only the location of waterbodies and no information on their dimensions (Fig. A1 b and c). To estimate the surface area of these systems, 100 water bodies were randomly selected using the

Subset Features tool in the Geostatistical Analyst toolbox in ArcGIS (Version 10.3, ESRI Inc., Redlands, California, USA). The surface area of selected water bodies was then quantified using high resolution aerial imagery (Nearmap; www.nearmap.com.au). Typical pixel resolution ~~was~~ of 7 cm~~, which~~ greatly improves edge detection of ponds as it can be very challenging to separate the water edge from riparian vegetation stands with coarser-scale data. Pond edges were mapped following the methodology of Albert et al., (2016) where imagery was georeferenced and the water edge was manually traced to create individual polygons for each pond. The mean surface area of all 100 polygons was then then assumed to approximate the surface area of all individual ponds within this database and the total surface area was calculated by multiplying this mean surface area by the total number of ponds.~~used to calculate total surface area of water bodies within this database.~~

To ensure only one water body was reported from each location, a~~A~~ll databases were first screened ~~to ensure only one water body was reported from each location, with overlapping~~ to remove repeated detections of waterbodies ~~removed~~. ~~The~~ All remaining water bodies were then summed together to calculate total surface area of ponds and this was compared to larger reservoirs to determine their relative surface area. To undertake regional scaling of pond emissions, individual ponds were sorted using two different size class classifications: Firstly, we categorised sites into the three smallest size classes (10^2 to 10^3 m^2; 10^3 to 10^4 m^2; and 10^4 to 10^5 m^2) in the Global Reservoir and Dam (GRanD) assessment (Lehner et al., 2011). Secondly, we divided sites into water bodies less than 3,500 m^2 (primarily stock dams) and larger water bodies (primarily irrigation dams and urban lakes), following the findings of Lowe et al., (2005) and SKM, (2012).

2.3 CH_4 emissions from broad spectrum of pond types

To quantify the range of emission rates from ponds, a monitoring program was undertaken from August to December 2017 across a wide spectrum of ponds including: farm dams (irrigation and stock watering), urban lakes, small weir systems (i.e. small dams leading to widening and slowing of river flows) and rural residential water supplies (Fig. 1). Stock dams, irrigation dams and urban lakes account for the vast majority of ponds across Queensland and ponds within each category were selected to represent the regional size class distribution (Fig. A2). The majority of sites were located in coastal catchments in south east Queensland, Australia as well as one urban lake and three stock dams in Central Queensland (Fig. 2 c).

There are a number of commonly used methods to assess methane emissions from water bodies depending on the pathway of interest. For the diffusive emission pathway, rates may be modelled using the thin boundary methods or directly measured using manual or automatic floating chambers (St. Louis et al., 2000). For ebullition pathways, rates can be directly measured using acoustic surveys or funnel traps (DelSontro et al., 2011). Thin boundary layer models cannot be used to quantify the ebullition pathway and acoustic surveys or funnel traps cannot be used effectively in ponds as the water depth is often too shallow (< 1 m). We chose to use floating chambers to capture both ebullition and diffusive fluxes. CH_4 emission rates were measured by deploying between 3 and 16 floating chambers per water body, ~~capturing~~ covering both peripheral and central zones (Fig. ~~A2~~A3). Chamber design followed the recommendations of Bastviken et al., (2015), as these lightweight chambers (diameter 40 cm, 12 L headspace volume and 0.7 kg total weight) were ideally suited to deployment in ponds where both site access and on-water deployments can be challenging (Fig. A4). The floating chambers used were designed to yield negligible bias on the gas exchange and compare well with non-invasive approaches (Cole et al., 2010;Gålfalk et al., 2013;Lorke et al., 2015).

Where possible 24 hour measurements were undertaken, however in three water bodies this was not possible, (Table A1) and here measurements lasted between 6 and 8 hours. The 24 hour deployment time was chosen to increase the likelihood to capture ebullition, which is episodic in nature, and to incorporate diel variability in diffusive emissions which can be up to a 2-fold bias (Bastviken et al., 2004;Bastviken et al., 2010;Natchimuthu et al., 2014). The use of long term deployments may underestimate diffusive fluxes, which decrease as the chamber headspace approaches equilibrium with the water. However, in contrast to CO_2, CH_4 has a long equilibration time and it has been shown that a 24 hour deployment of these types of flux chambers on lakes underestimate diffusive fluxes by less than 10% (Bastviken et al., 2010). An initial gas sample was collected at chamber deployment and a final ~~After each deployment a~~ chamber headspace gas sample after 24 hours ~~was collected~~ following the Exetainer method described in Sturm et al., (2015). CH_4 emission rates were calculated from the change in headspace concentration over time and normalised to areal units (Grinham et al., 2011).

2.4 ~~Spatial and temporal v~~Variability in surface area and emission rate

2.4.1 Spatial and seasonal variability across a single water body

To gain insight into the spatial and temporal uncertainty in pond emissions we compared variability in seasonal emissions from a single site to emissions from an intensive spatial survey of multiple sites across the pond (Fig. 4). Seasonal variability in emission rates ~~were~~ was measured at an urban lake (St Lucia 1) where monthly monitoring at a single site was undertaken across an annual cycle (Jan to Dec 2017). This pond was selected as water level remains relatively constant throughout the year and sampling would not impacted by changes in inundation status. Emissions were monitored following the same methodology as described in the preceding section, and 4 or~~to~~ 5 floating chambers were deployed for each sampling event. Emission rates from this seasonal study were then compared to an intensive spatial survey of the same ~~lake~~pond (Dec 2017), where 16 chambers were deployed simultaneously ~~across the lake~~for a 24 hour incubation. To better understand spatial patterns in emissions within this pond the water depth and proximity to inflow points were mapped. The bathymetric survey was conducted using a logging GPS depth sounder (Lowrance HDS7 depth sounder, Navico, Tulsa, Oklahoma, USA). Georeferenced water depth points were imported into ArcGIS and interpolated across the whole water body using the inverse distance weighting function.

2.4.2 Variability in water surface area across all monitored ponds

The variability in surface area of each of the 22 ponds monitored in the emissions surveys was analysed ~~The variability in surface area of each pond monitored in the emissions surveys was analysed~~ using high resolution historical imagery across all monitored water bodies. A time-series of high resolution aerial imagery over a 9 year period from 2009 to 2017 was screened for image quality and appropriate images were selected. The time series data are not consistent across the whole state, the number of discrete images for individual water bodies varied from 3 to 16. Images of individual ponds were georeferenced to a common permanent feature across all images and then the outer water edge was mapped and surface area calculated following Albert et al., (2016). The time series of surface area for individual water bodies was compared to their corresponding surface area at full supply level (A_{FSL}) and expressed as a percentage then grouped into three size classes based on the GRanD classification. This time period also captured the range of rainfall variability across the state with 2010 being the wettest year on record whilst 2013 to 2015 were consecutive drought years (Average rainfall; https://data.qld.gov.au/).

2.5 Effect of inundation status on pond emissions

~~Given the relatively shallow nature of most ponds, as well as high water use rates, peripheral areas of the water body regularly experience periods of inundation and no inundation as water levels change.~~ The effect of inundation status on emission rates was tested on a stock dam (Gatton 4) where measurements were undertaken on peripheral areas during periods of inundation and no inundation. This pond was selected as stock dams generally experience accelerated rates of water level change due to their relatively small size compared to other pond types (Fig. A2). In addition, the construction of this pond is typical for stock dams (a shallow pit is dug out and the soil used to construct the wall and spillway) and the surface area (1,893 m^2) closely matched the median for all farm dams (1,586 m^2; Fig. A2). Emission measurements for the inundated period followed the methodology outlined above for the water body emissions survey. Three weeks later water levels within the ponds had dropped and emission measurements were repeated at the same sites which were now exposed. For these emission measurements five chambers (90 mm diameter, 150 mm length) were carefully inserted 50 mm into the ground and care was taken to minimise disturbance to the soil surface. The headspace of each chamber was flushed with ambient air to remove headspace contamination due to chamber insertion, then the sampling port of each chamber was sealed. After the deployment period, a gas headspace sample was collected and CH_4 concentration analysed.

2.6 Statistical analyses and regional scaling of emissions

Emissions rates and surface area data were analysed using a series of one-way analyses of variance (ANOVAs) with the software program, Statistica V13 (Dell Inc., 2016). Analysis of emissions rates collected during the monthly monitoring study and the inundation study used sampling month or inundation status as the categorical predictors and chamber emission rates as the continuous variable. Emission rates from individual water bodies collected during the broad survey were first pooled into four primary use categories (irrigation, stock, urban and weirs) or three different GRanD size classes and these categories were used as the categorical predictors. The primary use of each pond was provided by pond owners or managers, in the case of urban lakes that had both aesthetic and stormwater functions these were classified as urban (Table A1). 22 ponds were included in this survey with four irrigation ponds, nine stock watering ponds, seven urban ponds and two weirs. Changes in water surface area (as a percentage of A_{FSL}) from individual water bodies were pooled into three GRanD size classes and these categories used as the categorical predictors. Where necessary, continuous variable data were log transformed to ensure normality of distribution and homogeneity of variance (Levene's test) with post hoc tests performed using Fisher's LSD (least significant difference) test (Zar, 1984). Tests for normality were conducted using Shapiro-Wilks tests as recommended by Ruxton et al., (2015). The non-parametric Kruskal–Wallis (KW) test was used for continuous data which failed to satisfy the assumptions of normality and homogeneity of variance even after transformation. Statistical results were reported as follows: Test applied (Fisher's LSD or Kruskal-Wallis test), the test statistic (F or H) value and associated degrees of freedom with p-value.

Emissions were scaled to water body size classes following two different approaches. Firstly, emissions were grouped according to their respective GRanD size class. These match the size class of water bodies used in the emissions monitoring of this study, and the GRanD database was used in the most recent global synthesis of greenhouse gas emissions from reservoirs (Deemer et al., 2016). Secondly, water bodies less than 3,500 m^2 in area were assumed to be primarily stock dams and larger water bodies primarily irrigation dams (Lowe et al., 2005). To extrapolate pond emission rates to regional scales, an appropriate measure of centrality should be used.

Three common measures, arithmetic mean, geometric mean and median values, were calculated for each water body category and size class. To assess the most suitable measure of centrality for water body emissions, normal probability plots of raw and log transformed emissions data were generated and tested using the Shapiro-Wilks test (Fig. A5). The emissions data from all replicate measurements fitted a log-normal ($p = 0.081$) but not a normal distribution ($p = 0.0000$)~~clearly followed a log normal distribution (Fig. A3)~~ and, therefore, the geometric mean would provide the most appropriate measure of centrality for this data (Ott, 1994;Limpert et al., 2001). Fluxes were scaled to annual rates using the cumulative surface area of water bodies and the respective emissions rate for each size class using the geometric means. The variability in geometric mean was given by the exponential of the 95% confidence interval range of log transformed data. Emissions for water bodies less than 3,500 m^2 were scaled using stock dam rates and larger water bodies (3,500 m^2 to 10^5 m^2) using rates obtained from irrigation dams and urban lakes. Total fluxes from respective size classes were then combined to provide regional estimates. Annual fluxes of CH_4 were converted to CO_2 equivalents assuming a one hundred year global warming potential of 34 (IPCC, 2013).

3 Results

3.1 Relative surface area of ponds

The state wide land use assessment identified 13,046 ponds across Queensland, occupying a total surface area of approximately 467 km^2 (Fig. 2 c). However, with the inclusion of the additional Reservoir and Water Storage Point datasets the number of ponds increased over 20 times to a total of 293,346, and the surface area more than doubled to 1,087 km^2. The official land use assessment of Queensland underestimates the surface area of ponds by 57%, and the total number of water bodies by more than an order of magnitude. The revised total surface area of all artificial water bodies across Queensland increased by 24% to just over 3,248 km^2 (Table A2).

Ponds were widely distributed across the state, but over 78% of ponds were located on grazing land, suggesting that stock dams represent the primary water body type (Fig. 2 a). ~~The majority~~Over two thirds of ponds were confined to regions of the state were rainfall isohyets were above 600 mm (Fig. 2 b) and heavily concentrated in cropping and residential areas in the central and south eastern parts of the state (Fig. 2 c). These findings highlight the importance of striving to incorporate all artificial water bodies into flooded land emission assessments; omitting water bodies below a size threshold can lead to a dramatic under-estimation of the total number of water bodies present, and a considerable underestimate of the available surface area for CH_4 emissions.

3.2 CH_4 emissions from ponds

All 22 water bodies monitored in this study were shown to be emitters of CH_4, and emission rates ranged from a minimum of 1 mg m^{-2} d^{-1} to a maximum of 5,425 mg m^{-2} d^{-1} (Table A1). Only one water body (Mt Larcom 3) had a maximum rate below the reported upper range (50 mg CH_4 m^{-2} d^{-1}) for diffusive fluxes found in larger water bodies in this region (Grinham et al., 2011;Musenze et al., ~~2016~~2014). Mean flux rates of only four individual water bodies were below 50 mg m^{-2} d^{-1} (Table A1) suggesting ebullition to be the dominant emission pathway in these systems.

Grouping ponds according to their primary use resulted in no significant differences in emissions rates between irrigation dams, stock dams and urban lakes, however, weirs were significantly higher ($F_{(3,121)} = 6.43$, $p < 0.001$)

than all other categories (Fig. 3 a). Mean emission rates were however higher in stock water bodies (168 mg m^{-2} d^{-1}) compared with irrigation and urban bodies (84 and 129 mg m^{-2} d^{-1}, respectively). Weir water bodies had mean emission rates of 730 mg m^{-2} d^{-1}, more than four times higher those of any other category (Fig. 3 a). Grouping ponds according to their GRanD size classes resulted in significantly higher emissions rates ($KW\text{-}H_{(2,121)} = 7.354$, $p < 0.05$) from ponds in 10^2 to 10^3 m^2 size class compared to 10^4 to 10^5 m^2 (Fig. 3 b). Overall, mean emissions decreased with increasing size class. Note that all weir sites fell into the smallest size category.

3.3 Spatial and temporal variability in surface area and emission rate

3.3.1 Spatial and temporal variability within a single pond

Observed emissions rates from the ~~high-resolution~~intensive spatial study, carried out in December 2017, ranged over two orders of magnitude from under 40 mg m^{-2} d^{-1} to over 3,500 mg m^{-2} d^{-1} (Fig. 4). Emissions were highest in the shallow southwest sector of the pond, adjacent a large stormwater inflow point, as well as along the western boundary where numerous overhanging riparian trees are located along with a second stormwater inflow point (Fig. 4).

Monthly emissions were moderately variable across the annual cycle and mean rates ranged from 176 to 332 mg m^{-2} d^{-1}. No significant difference in emissions rates ($KW\text{-}H_{(11,50)} = 3.56$, $p = 0.98$) were observed between sampling events (Fig. 5). Mean rates observed during the monthly monitoring were similar to chamber rates from the intensive spatial study (274 mg m^{-2} d^{-1}).

3.3.2 Variability in water surface area across all monitored ponds

Variability in water surface area is strongly related to water body size class (Fig. 6). Mean surface area within the smallest size class was only 64% of A_{FSL}, this increased to 84% in the intermediate size class and ~~was~~ to 94% in the largest size class (Fig. 6). Smaller ponds had a significantly lower surface area relative to A_{FSL} ($KW\text{-}H_{(2,231)} = 50.523$, $p < 0.001$) compared to larger size classes and were more variable (Fig. 6). Regional emissions estimates therefore need to correct for the differences in water body surface area relative to predicted A_{FSL}, particularly, in the smaller size classes.

3.4 Effect of inundation on stock dam~~pond~~ emissions

The water surface area of a single stock dam ranged from 395 to 2,808 m^2 over a 40 month period (Fig. 7 a) with an outer band of 580 m^2 undergoing frequent inundation cycles (May 2016 to Dec 2017 - Fig. 7 a). Emissions rates from peripheral areas during an inundated period were significantly higher (more than one order of magnitude) compared with emissions when not inundated ($KW\text{-}H_{(1,10)} = 6.818$, $p < 0.001$; Fig. 7 b). In contrast emissions from central areas were over 100 mg m^{-2} d^{-1}, more than double the peripheral area emission rates (Table A1). This modifier of rates will primarily impact emissions from smaller size classes which have greater variability in water surface area (Fig. 6). An additional implication is in the importance of designing monitoring studies where emissions rates are quantified from both peripheral and central areas for each system. Rates monitored only in peripheral areas will likely bias towards lower emissions, particularly if these have undergone recent inundation.

4 Discussion

4.1 Relative importance of pond emissions to regional flooded land inventories

The findings of this study ~~clearly~~ demonstrate ponds are an underreported and important CH_4 emission source in Queensland, and likely also globally. ~~The official land use assessment of Queensland underestimates the surface area of ponds by 57%, and the total number of water bodies by more than an order of magnitude.~~ These findings highlight the importance of striving to incorporate all artificial water bodies into flooded land emission assessments; omitting water bodies below a size threshold can lead to a substantial under-estimation of the total number of water bodies present, and a considerable underestimate of the available surface area for CH_4 emissions. ~~The revised total surface area of artificial water bodies across Queensland increased by 24% to just over 3,248 km^2 (Table A2).~~ Mean annual CH_4 fluxes from ponds for the State of Queensland ranged between 1.7 and 1.9 million t CO_2 eq (Table 1) depending on the scaling approach. Given ponds represent 33.5% of the total flooded lands surface area in Queensland and emission rates are equivalent to larger water bodies in the region (Musenze et al., 2014;Sturm et al., 2014), ponds represent one-third of total emissions from flooded lands in Queensland.~~The uncertainty in mean emissions ranged from a lower limit of 1.1 million t CO_2 eq to an upper limit of 3.2 million t CO_2 eq.~~ Remarkably, mean total emissions from ponds represent approximately 10% of Queensland's land use, land use change and forestry sector (NGERS, 2015) emissions using either scaling approach.

Future regional and global emissions estimates would be greatly improved with the inclusion of ponds, as their proliferation has been noted in five continents. In the continental United States ponds have been shown to cover 20% of total artificial water body surface area (Smith et al., 2002); in South Africa there are an estimated 500,000 ponds (Mantel et al., 2010); in Czechoslovakia ponds make up over 30% of total artificial water bodies surface area (Vacek, 1983); and in India ponds are estimated to comprise 6,238 km^2, or over 25% of India's artificial water body surface area (Panneer Selvam et al., 2014).

4.2 Pond emission pathways

Emissions rates from ponds observed in this study are consistent with ebullition being the dominant pathway. Diffusive emissions from studies of three larger water bodies in the region found the upper limit for diffusive emission was 50 mg m^{-2} d^{-1} (Grinham et al., 2011;Musenze et al., ~~2016~~2014) and only five ponds had emission rates below this level. Ebullition was observed at all ponds with maximum rates all in excess of 50 mg m^{-2} d^{-1} with the exception of only one stock dam (Mt Larcom 3) where the maximum rate was 19 mg m^{-2} d^{-1}. This is a consistent finding with larger water bodies in the region where ebullition has been shown to dominate total emissions (Grinham et al., 2011;Sturm et al., ~~2016~~2014). The relatively higher emissions from smaller pond size classes is consistent with previous observations of increased ebullition activity in shallow zones, particularly water depths less than 5 m (Keller and Stallard, 1994;Joyce and Jewell, 2003;Sturm et al., ~~2016~~2014). Virtually all ponds within the smaller size classes would be less than 5 m deep. In addition, ponds trap large quantities of sediment and organic material (Neil and Mazari, 1993;Verstraeten and Prosser, 2008) and these deposition zones have been identified as methane ebullition hotspots in larger water bodies (Sobek et al., 2012;Maeck et al., 2013). The pattern in emissions from the intensive spatial study in an urban lake, where shallow areas adjacent stormwater inflows were shown to be ebullition hotspots, have also been observed in larger water bodies were ebullition activity was highest adjacent to catchment inflows (DelSontro et al., 2011;Grinham et al., 2017;de Mello et al., 2017). The emissions from small weirs were clearly dominated by ebullition, which is consistent with emissions from three larger weirs where rates ranged from 1,000 to over 6,000 mg m^{-2} d^{-1} (Bednařík et al., 2017). Weirs intercept the primary streamflow pathways and will likely cause large quantities of catchment derived organic matter to deposit within the weir body which, coupled to the shallow nature, results in high rates of ebullition. Overall, the rates observed for all categories, except irrigation dams, were in the upper range of reservoir areal flux rates reported in global reviews (St. Louis et al., 2000;Bastviken et al., 2011;Deemer et al., 2016), reflecting the dominance of the ebullition pathway in ponds. An additional consideration for future studies of ebullition patterns in ponds stems from recent studies of reservoirs which found significant changes in ebullition intensity and ebullition distribution as water levels decrease (Beaulieu et al., 2018;Hilgert et al., 2019). Under decreasing water levels, deeper zones of ponds may begin bubbling or increase the intensity of bubbling, this could potentially offset the reduction in surface available for emissions and total emissions would remain relatively constant.

4.3 Challenges in scaling emissions

Efforts to develop flooded land emission inventories rely heavily on the emission rate used to scale the surface area of water bodies' within selected categories. Given the high variability in emission rates within and between individual ponds and relatively low replication, it is critical to select an appropriate measure of centrality (arithmetic mean, geometric mean or median) in order to scale regionally and globally (Downing, 2010). For rice paddies, septic tanks, peatlands and natural waters (Aselmann and Crutzen, 1989;Dise et al., 1993;Diaz-Valbuena et al., 2011;Bridgham et al., 2006) the geometric mean has been applied. Likewise, in this study the log normal distribution of emissions data indicated the geometric mean as the most appropriate measure and the total emission rates using this measure fell within the reported range from larger artificial water bodies in the region (Grinham et al., 2011;Sturm et al., ~~2016~~2014). However, the geometric mean for all water body categories and size classes were less than half of their respective arithmetic mean values (Fig. ~~A4~~A6). For irrigation, stock and urban water bodies, geometric mean values were actually outside of 95% confidence interval limit for the arithmetic mean (Fig. ~~A4~~ A6 a and b). Geometric mean and median values were similar across all water body categories and size classes and these measures, therefore, represent conservative emissions rates from ponds. This raises an important issue with scaling ebullition dominated water bodies as there is always going to be a high likelihood of detecting a small number of very high rates which will invariably give rise to log normal data distributions. Future studies will focus on determining whether the conservative estimates generated through the use of geometric means approximate the true emissions from ponds.

5 Future research

Continued efforts to quantify regional pond abundance, particularly smaller size classes, should be a research priority as this will greatly improve the surface area estimate of flooded lands used for upscaling greenhouse gas emissions as well as their role in the global carbon cycle.~~will greatly improve flooded land surface area estimates available for emissions.~~ The increased coverage, availability and resolution of satellite imagery as well as more sophisticated methods to identify water bodies (Verpoorter et al., 2014) will support these efforts. However, it is critical to continually update regional assessments as the annual increase in farm ponds has been estimated to be as high as 60% in some parts of the globe (Downing and Duarte, 2009). Regional assessments should also correct for differences in pond surface area, particularly in the smaller size classes, as this study has demonstrated actual surface area can be significantly smaller than the surface area at full supply level~~,~~ (A_{FSL}). An additional consideration is to ensure pond emission studies from different regions include all relevant ponds types. For example, the use of ponds to increase groundwater recharge is widespread across South East Asia (Giordano, 2009) and these would need to be included in regional inventories.

Increasing both the number and type of pond within each size class in emissions monitoring studies should be a research priority~~ for emissions monitoring studies~~. This will allow increased confidence in the selection of an appropriate measure of centrality as well as reducing uncertainty in the expected range of emission rates within each pond category. When designing a monitoring study it is important to ensure emissions rates are quantified from both peripheral and central areas for each pond. This study demonstrated that measurements taken only in peripheral areas will likely bias towards lower emissions particularly in ponds that experience rapid changes in water level and, therefore, inundation status of peripheral areas. However, this was limited to a single stock dam and additional pond types and size classes must be examined before more confident generalisations can be made. The high spatial variability in emission rates within ponds noted from this study, highlights the importance of ensuring chambers cover the widest possible spatial scale during a measurement campaign. This will increase the likelihood of detecting ebullition zones which are likely the dominant emission pathway. However, this finding was from a single urban lake and additional long term temporal studies along with high resolution spatial surveys of different pond types and size classes are required to identify the drivers of pond emission pathways. Research into both pond surface area and CH_4 emission rates will allow greater understanding of their importance to flooded land emission inventories at both regional and global scales.

Data Availability

The data that support the findings of this study are available from the corresponding author upon request.

Author contribution

AG conceived, designed and conducted the study and co-wrote the manuscript; CE, CL, DB and BS conceived, designed the study and contributed to the manuscript; SA, ND and MD conducted the study and contributed to the manuscript.

Competing interests

The authors declare no competing financial interests.

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Tables

Table 1. Summary of Queensland small water bodies classified using two different relative size classifications. The number of water bodies, corrected surface area of size class and total mean annual emissions. Approach 1: emissions for water bodies less than 3,500 m^2 were assumed to be stock dams and larger water bodies irrigation dams (Fig. 3 a), Approach 2: emissions for GRanD size classes were taken from Figure 3 b, however, weir emissions were omitted as these are not relevant at the regional scale.

Approach 1					
Water body size (m^2)	**Number**	**Surface area (km^2)**	**Total emissions (t CO_2 eq yr^{-1})**		
			Mean	*Lower limit*	*Upper limit*
< 3,500	227,397	243	507,633	278,205	926,267
3,500 to 10^5	65,949	844	1,158,069	782,244	1,714,458
Total	**293,346**	**1,087**	**1,665,702**	**1,060,448**	**2,640,725**
Approach 2					
Water body size (m^2)	**Number**	**Surface area (km^2)**	**Total emissions (t CO_2 eq yr^{-1})**		
			Mean	*Lower limit*	*Upper limit*
10^2 to 10^3	108,526	50	97,302~~241,262~~	35,436~~112,316~~	267,177~~518,243~~
10^3 to 10^4	163,803	400	868,201	513,740	1,467,225
10^4 to 10^5	21,017	637	759,247	462,561	1,246,228
Total			**1,724,749~~1,868,710~~**	**1,011,736~~1,088,617~~**	**2,980,629~~3,231,695~~**

Figures

Figure 1. Oblique drone images showing examples of ponds where CH_4 emissions were monitored during this study: a) urban lake (St Lucia 1); b) stock dams in foreground (including Gatton 4), irrigation dam in background; c) small weir showing high organic loading upstream of wall (Mt Cootha); d) rural residential dam (Greenbank).

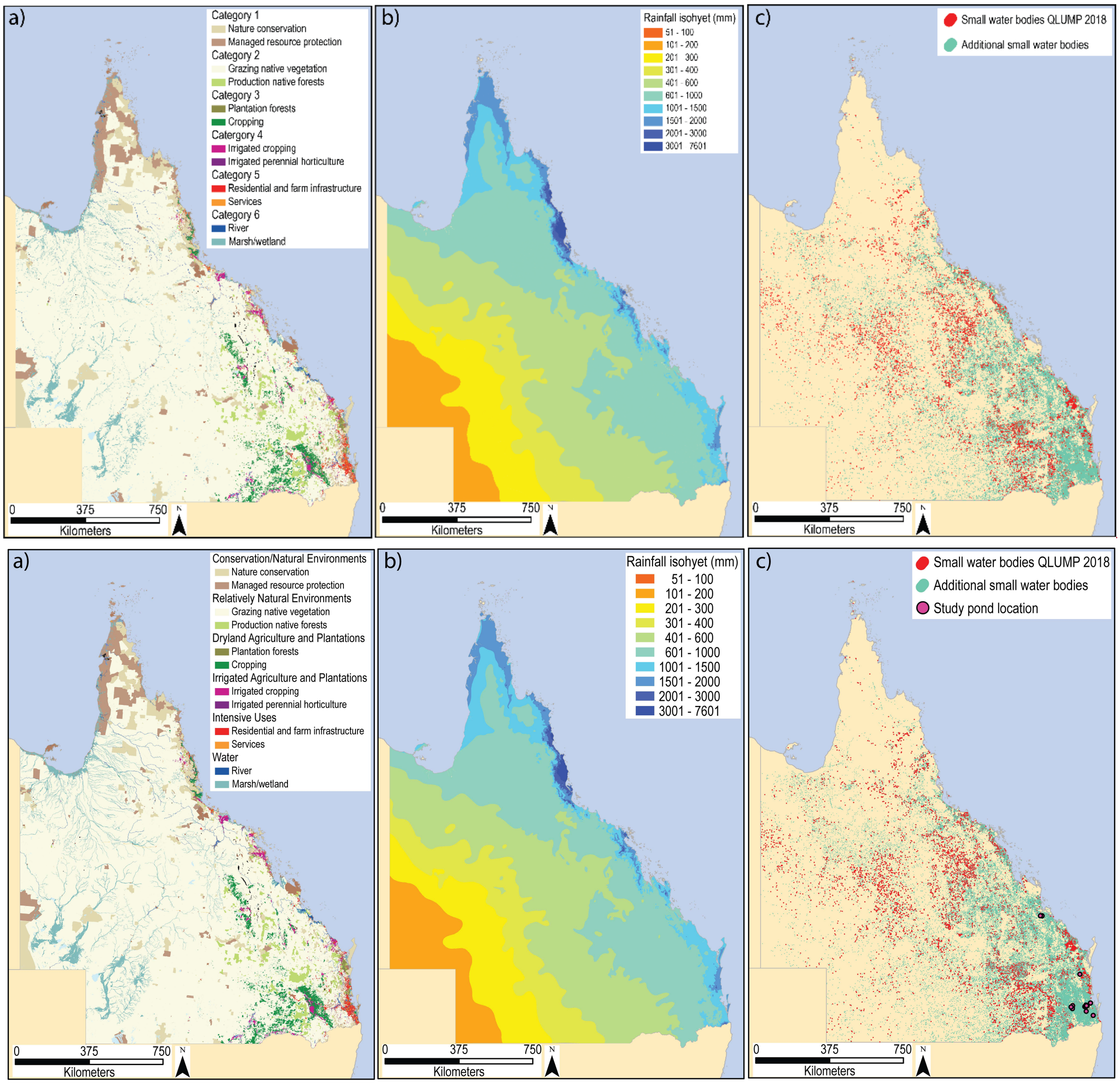


Figure 2. a) 2018 state wide assessment showing the relative surface area occupied by secondary land use categories (QLUMP, 2018). Note the legend shows the two largest land uses within each category.~~: Category 1 is Conservation and Natural Environments; Category 2 is Production from Relatively Natural Environments; Category 3 is Production from Dryland Agriculture and Plantations; Category 3 is Production from Dryland Agriculture and Plantations; Category 4 is Production from Irrigated Agriculture and Plantations; Category 5 is Intensive Uses; Category 6 is Water.~~ b) Mean annual rainfall isohyets across Queensland from 30 period of 1961 to 1990 (http://www.bom.gov.au accessed March 2018). c) Location of study ponds and ponds identified from the land use assessment (QLUMP 2018) and two additional state wide ~~assessments~~ databases (see text).

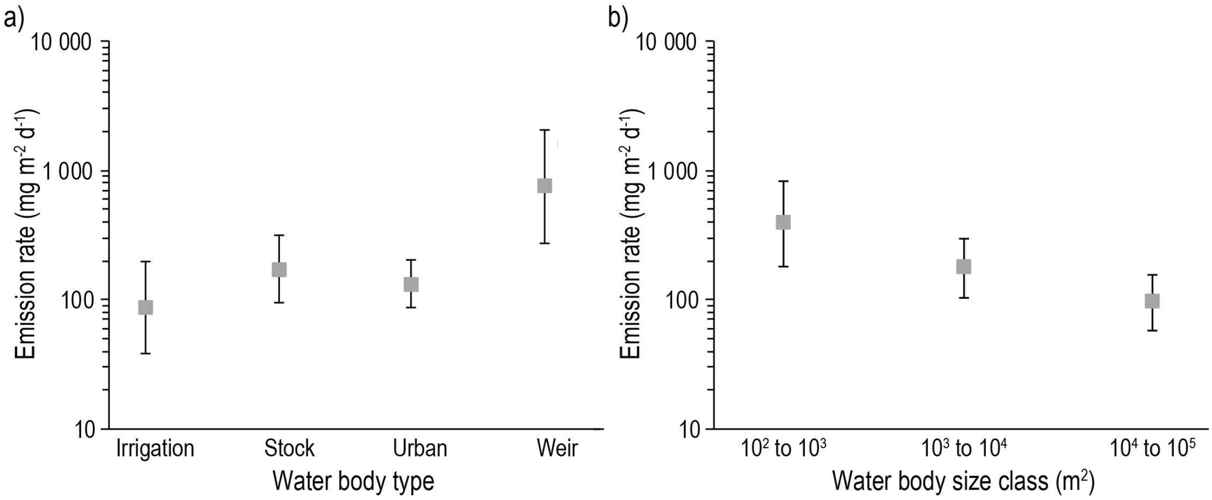


Figure 3. Mean CH_4 emissions across a) four categories of small water bodies (irrigation dams, stock dams, urban lakes and weirs) and b) three GRanD water body size classes. Values indicate geometric mean emission rates and 95% confidence intervals (± 95% CI).

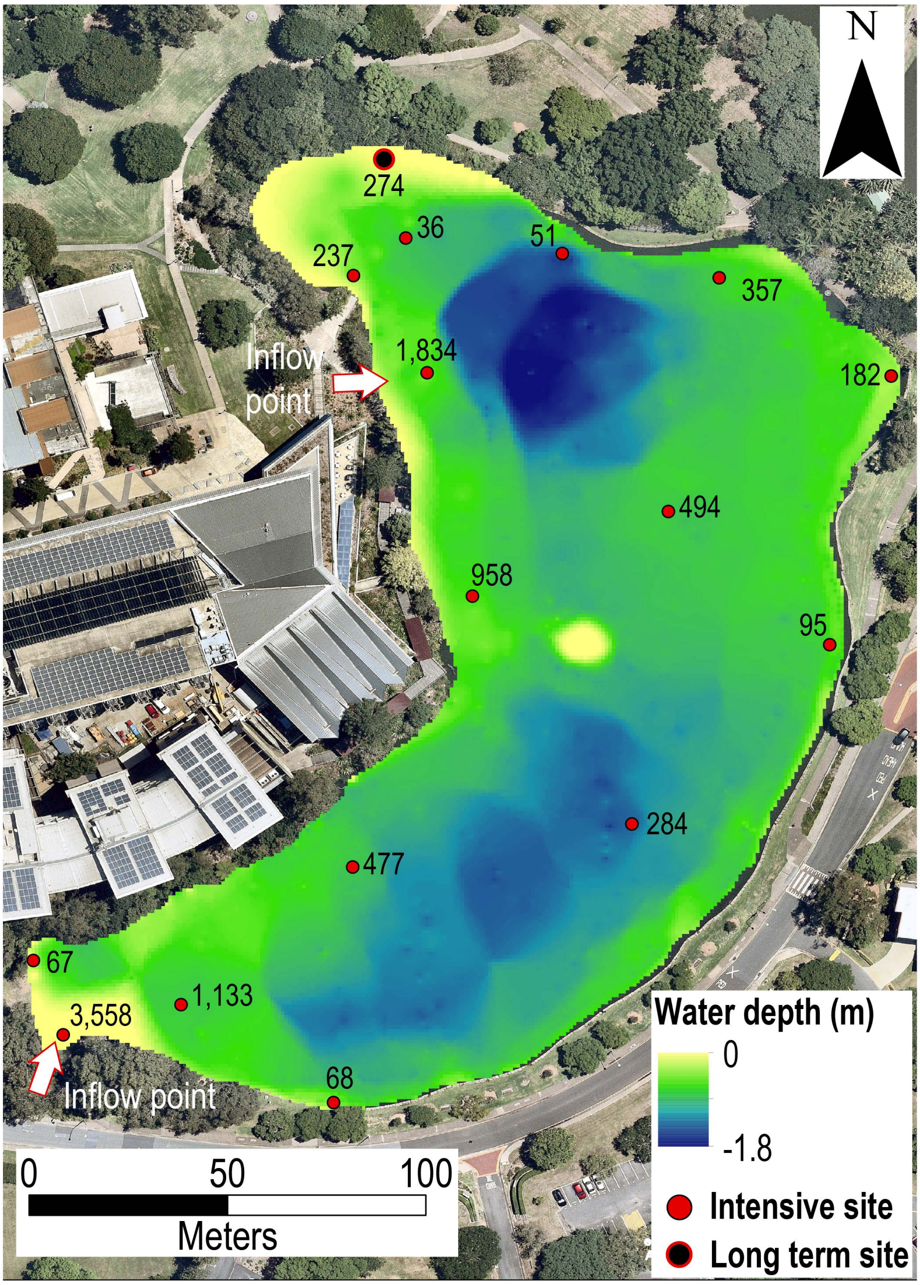


Figure 4. Sampling site location and chamber emission rates (mg m^{-2} d^{-1}) across an urban lake (St Lucia 1) relative to water depth and proximity to stormwater inflow points.

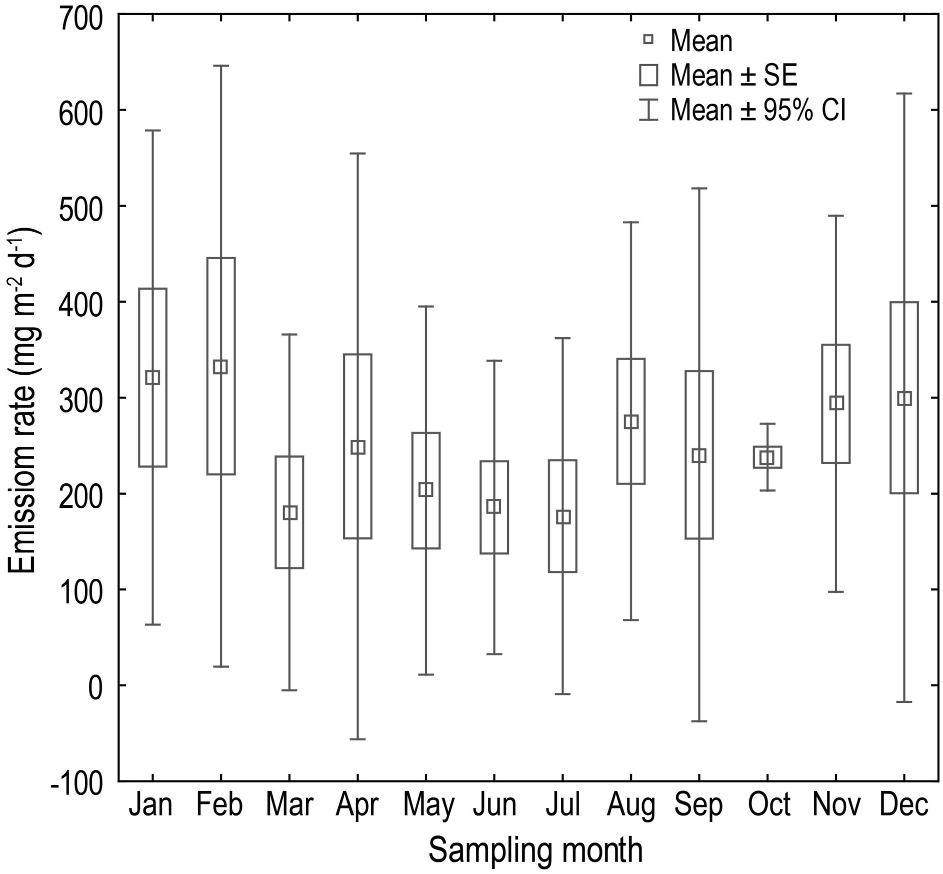


Figure 5. Monthly CH_4 emissions from a single monitoring site on an urban lake (St Lucia 1) across the annual cycle. Values indicate mean emission rates ± SE (standard error) and 95% CI (confidence intervals).

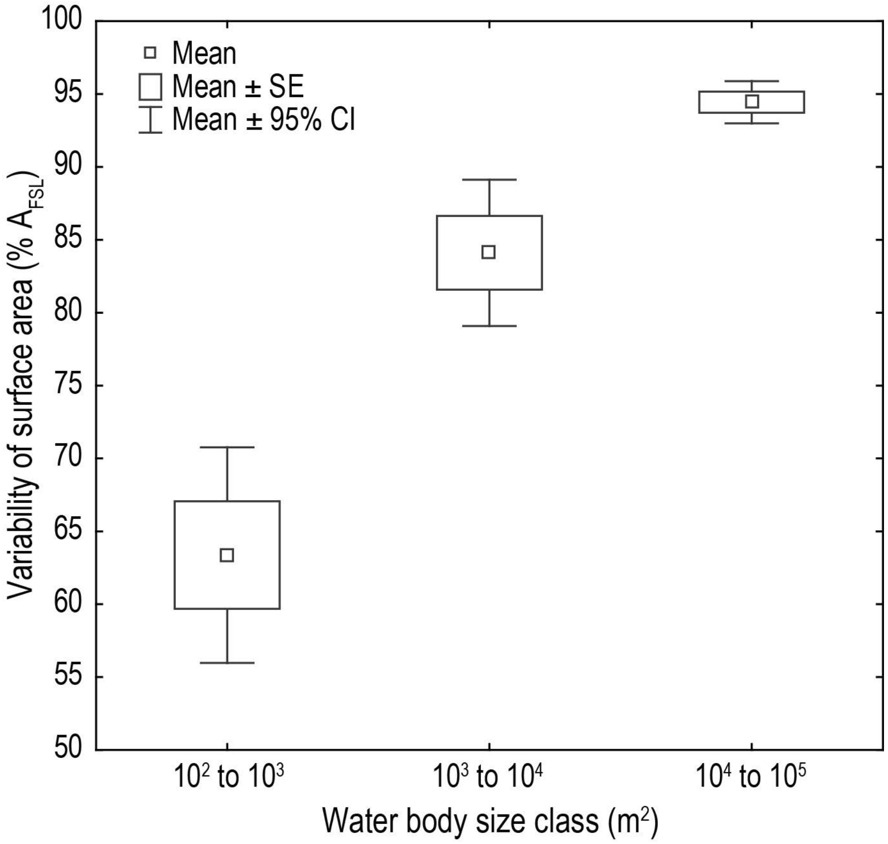


Figure 6. Variability in water surface area as a percentage of A_{FSL} between three GRanD database size classes of ponds. Values indicate mean surface area ± SE (standard error) and 95% CI (confidence intervals).

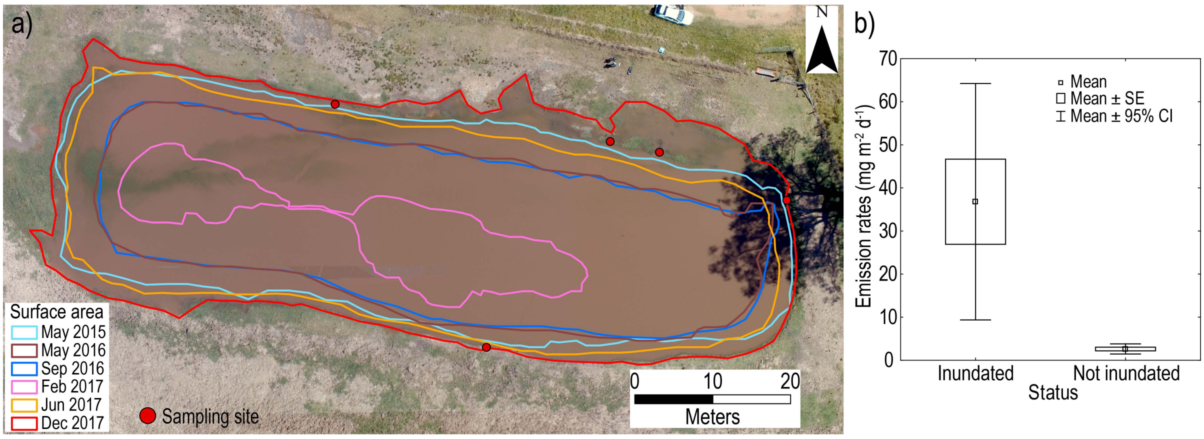


Figure 7. a) Changes in surface area of stock dam (Gatton 4) over a 40 month period. b) Emissions rates from peripheral zones during a period of inundation and no inundation. Values indicate mean emission rate ± SE (standard error) and 95% CI (confidence intervals).

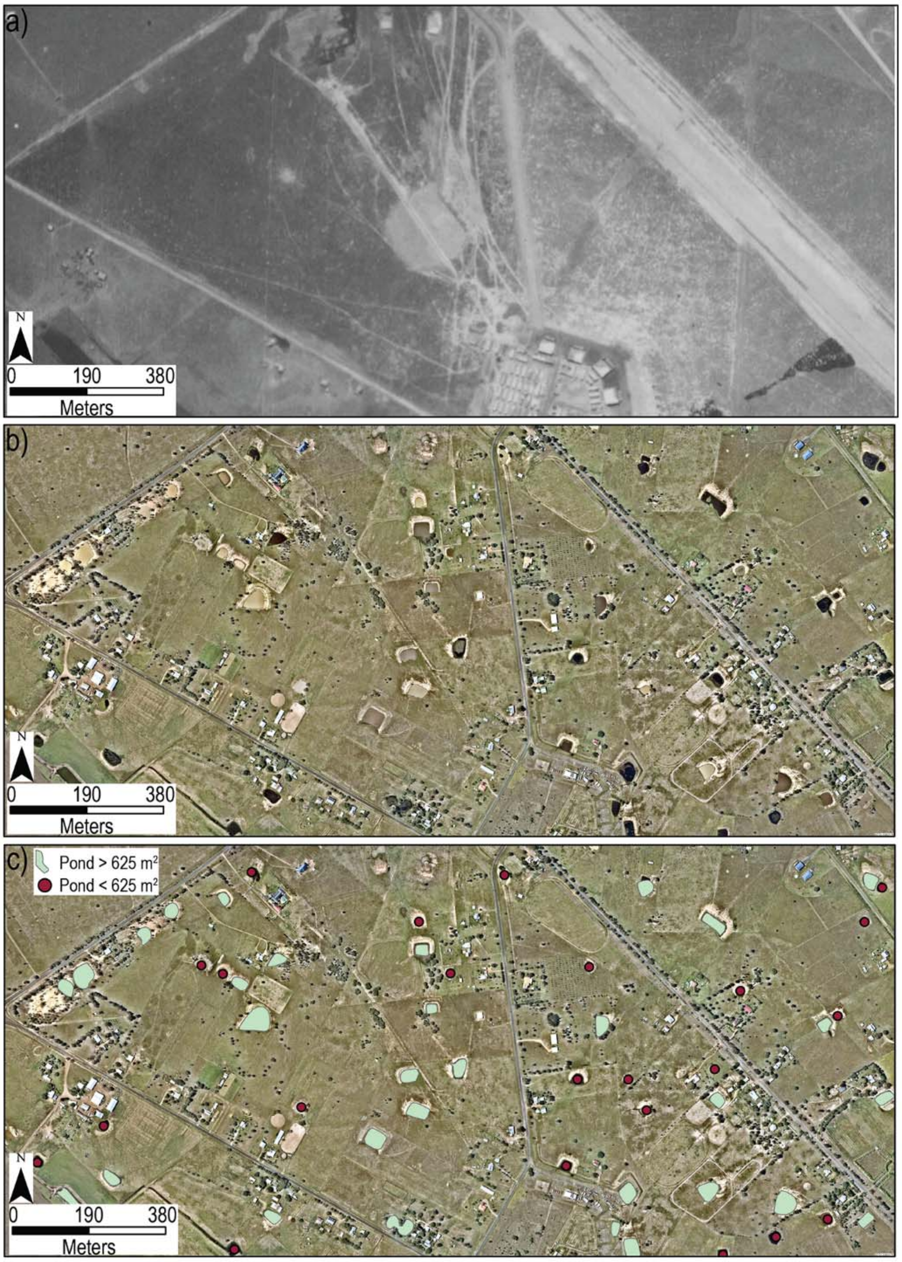
Appendix

Table A1: Selected characteristics from individual ponds showing: primary use of each system; surrounding land-use type; location of system latitude (Lat) and longitude (Long); average surface area (SA) in m^2; mean, median, minimum (Min) and maximum (Max) methane emission rates (mg m^{-2} d^{-1}); number of chamber measurements on individual systems (Cham). Primary uses included the following: irrigation for cropping; stock watering for cattle and horses; urban uses included stormwater management and aesthetic purposes; weirs for water supply and stream flow monitoring. * indicates water bodies where repeat sampling was conducted; # indicates water bodies where deployments of less than 24 hours were conducted.

Area	Primary use	Land-Use	Lat	Long	SA	Arth Mean	Geo Mean	Median	Min	Max	Cham
Gatton 1*	Irrigation	Grazing	-27.5541	152.3412	25,903	785	590	527	238	1,648	6
Gatton 2*	Irrigation	Grazing	-27.5548	152.3394	3,450	581	170	140	17	2,261	6
Gatton 3*	Stock	Grazing	-27.5615	152.3434	1,041	1,149	905	980	314	2,007	12
Gatton 4*	Stock	Grazing	-27.5625	152.3447	1,893	63	55	63	20	109	6
Gatton 5	Irrigation	Cropland	-27.5537	152.3503	30,458	129	122	110	89	186	3
Gatton 6	Stock	Cropland	-27.5546	152.3488	446	1,229	724	844	93	3,635	6
Port precinct#	Urban	Settlement	-27.3917	153.1676	38,285	144	57	68	8	357	3
St Lucia 1*	Urban	Settlement	-27.4996	153.0163	22,727	632	282	279	36	3,558	16
St Lucia 2	Urban	Settlement	-27.4984	153.0173	4,291	92	83	76	51	148	3
St Lucia 3	Urban	Settlement	-27.4981	153.0167	1,755	56	49	43	27	115	5
Pinjarra 1*	Irrigation	Grazing	-27.5372	152.9139	56,782	34	15	20	2	122	10
Pinjarra 2	Stock	Grazing	-27.5294	152.9242	1,943	205	59	277	2	335	3
Pinjarra 3	Stock	Grazing	-27.5294	152.9227	210	193	143	107	67	404	3
Oxenford	Urban	Settlement	-27.8924	153.2997	36,938	97	94	81	76	133	6
Mt Larcom 1	Stock	Grazing	-23.8008	150.9558	5,025	574	37	18	1	2,051	5
Mt Larcom 2	Stock	Grazing	-23.806	150.9574	1,256	48	45	49	26	70	3
Mt Larcom 3	Stock	Grazing	-23.8015	150.9446	16,093	17	17	18	14	19	3
Fig Tree Park	Urban	Settlement	-27.5394	152.9682	8,357	709	301	289	19	1,850	5
Greenbank#	Stock	Settlement	-27.7249	152.9779	575	290	166	188	29	755	4
Lake Alford#	Urban	Settlement	-26.2152	152.6848	21,689	49	29	62	5	79	3
Mt Cootha*	Weir	Forest	-27.4763	152.9642	580	2,493	1,405	2,337	368	5,425	6
Indooroopilly	Weir	Settlement	-27.5027	152.988	436	413	274	314	77	947	4

Table A2: Surface area (SA) of Queensland artificial water bodies within each GRanD database size class showing the official land use assessment estimates (QLUMP, 2018) and the revised estimates for the smallest three size classes found in this study.

GRanD size class (m^2)	QLUMP SA (km^2)	Revised SA (km^2)
10^2 to 10^3	0.005	50.3
10^3 to 10^4	8.4	400
10^4 to 10^5	459	637
10^5 to 10^6	605	605
10^6 to 10^7	555	555
10^7 to 10^8	553	553
10^8 to 10^9	448	448
Total	**2,629**	**3,248**


a)
N
0 190 380
Meters
b)
N
0 190 380
Meters
c)
Pond > 625 m²
Pond < 625 m²
N
0 190 380
Meters

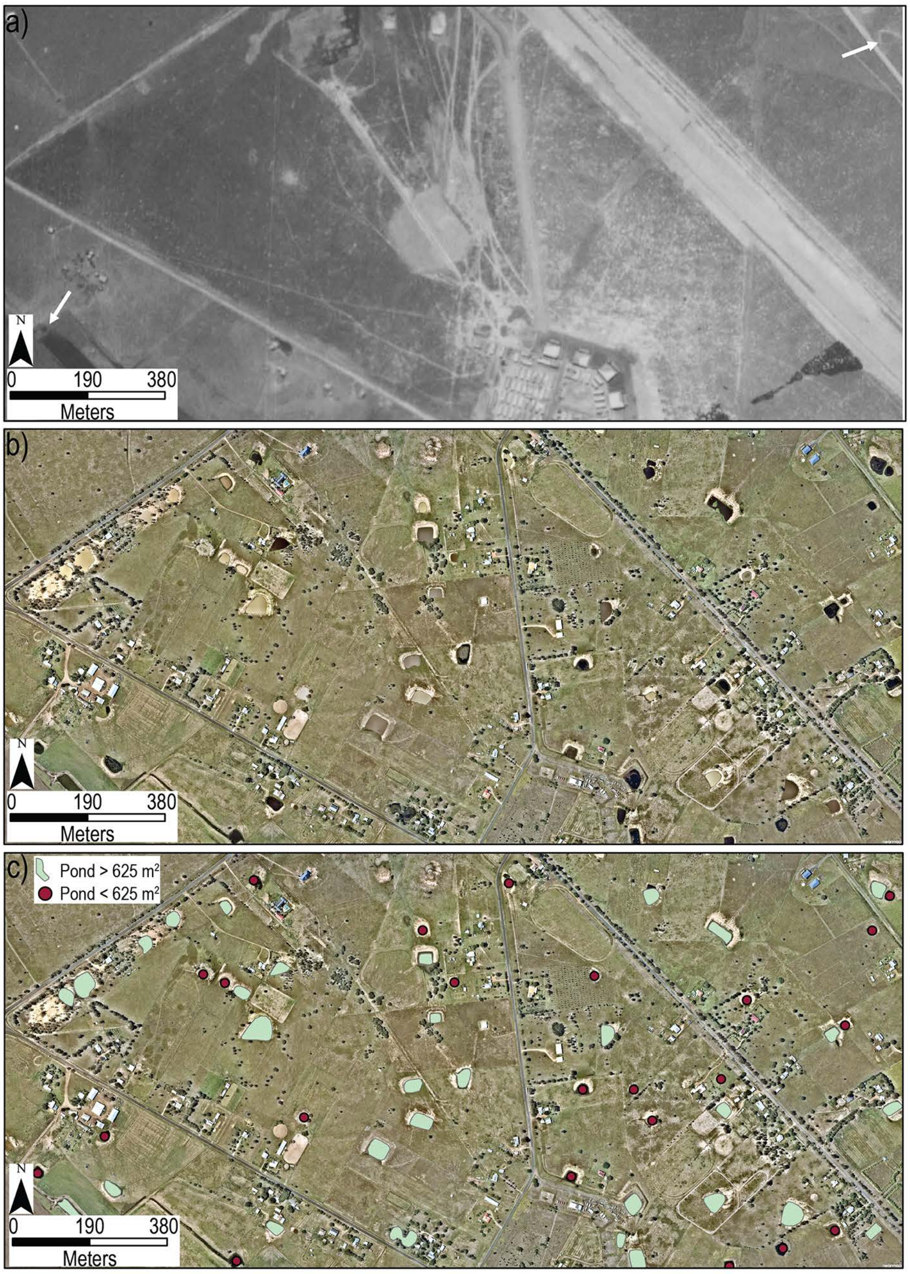


Figure A1. Historical changes in pond distribution from a 2.7 km^2 area in ~~South~~ south ~~East~~ east Queensland, Mt Tarampa (27°27'44"S, 152°28'59"E). a) 1944 aerial images showing 2 ponds indicated by white arrows, b) 2017 aerial image showing 54 ponds and c) showing the relative distribution of ponds from Reservoir (>~~-~~625 m^2~~-~~) database and Water Storage Point (<~~-~~625 m^2~~-~~) database, and together this results in a density of 20 ponds km^{-2}.

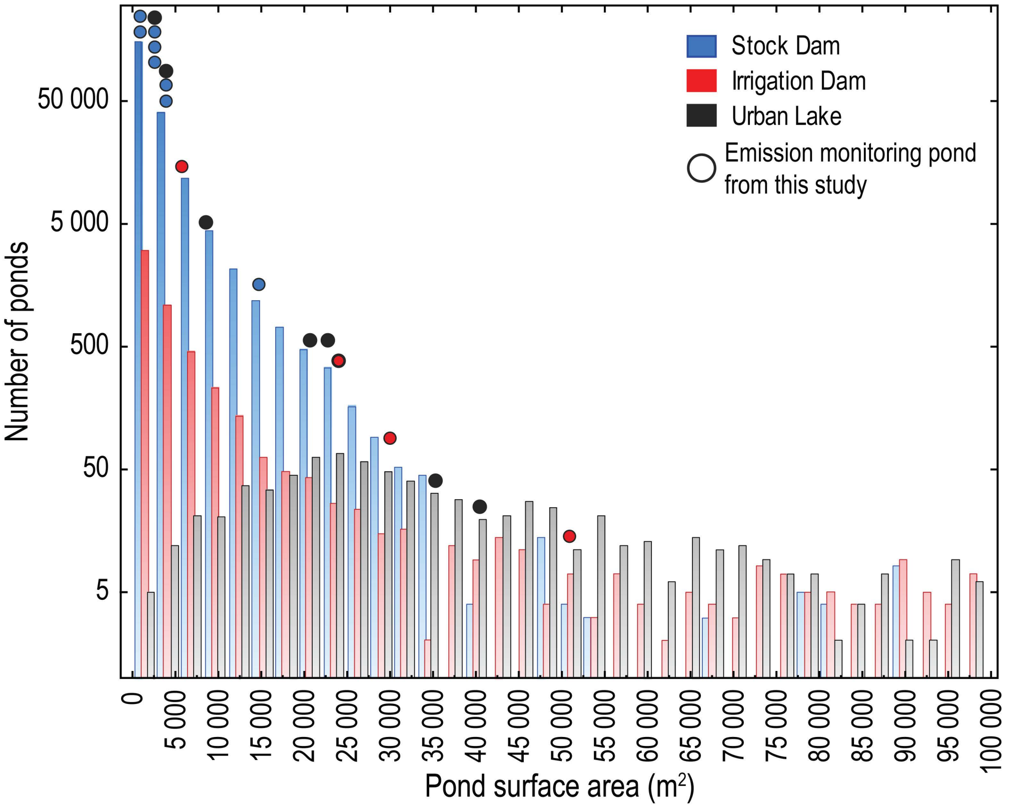


Figure A2. Pond size from emission study relative to histogram of regional pond distribution of stock dams, irrigation dams and urban lakes. The surface area of pond used emission study (Table A1). Histogram of regional distribution of ponds was developed from QLUMP, Reservoir and Water Storage Points databases and separated into pond type depending on surrounding land use: "Grazing native vegetation" for stock dams; "Production from irrigated agriculture and plantations" for irrigation dams; "Intensive uses" for urban lakes with "Mining" and "Manufacturing" landuse within "Intensive Uses" were removed to ensure only urban areas were selected. To incorporate the distribution of ponds within the Water Storage Points database, it was assumed this would match the distribution from the 100 individual ponds examined in Section 2.2 to determine their average surface area.

Figure ~~A2~~A3. An oblique drone image showing a nine floating chamber deployment setup targeting peripheral and central zones on a stock watering dam (Gatton 3).

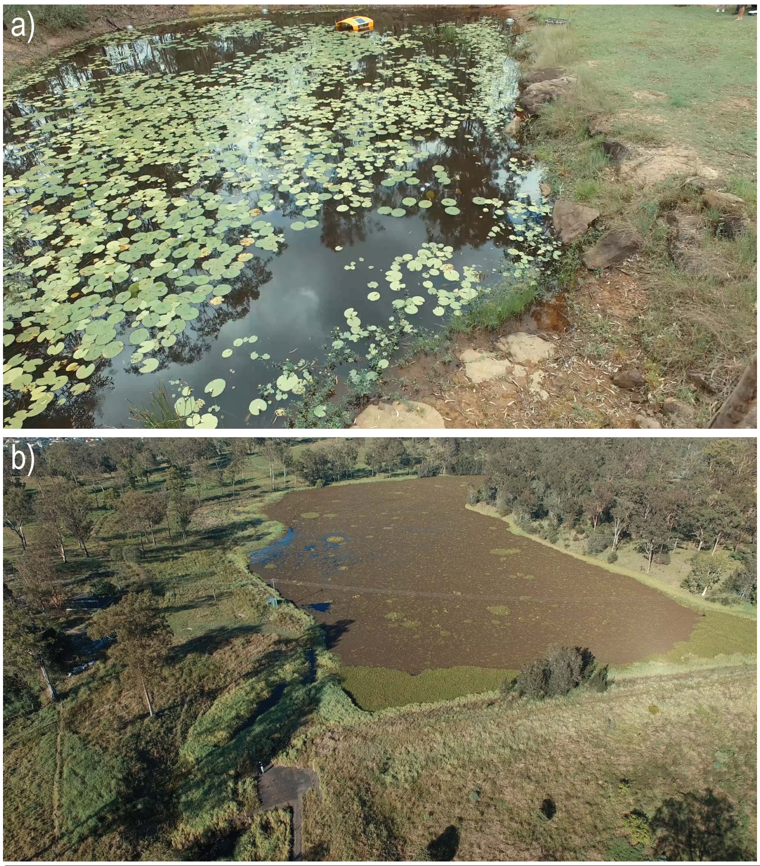


Figure A4. Oblique drone images showing natural obstacles for pond chamber deployments from a) emergent macrophytes and b) floating aquatic weeds.

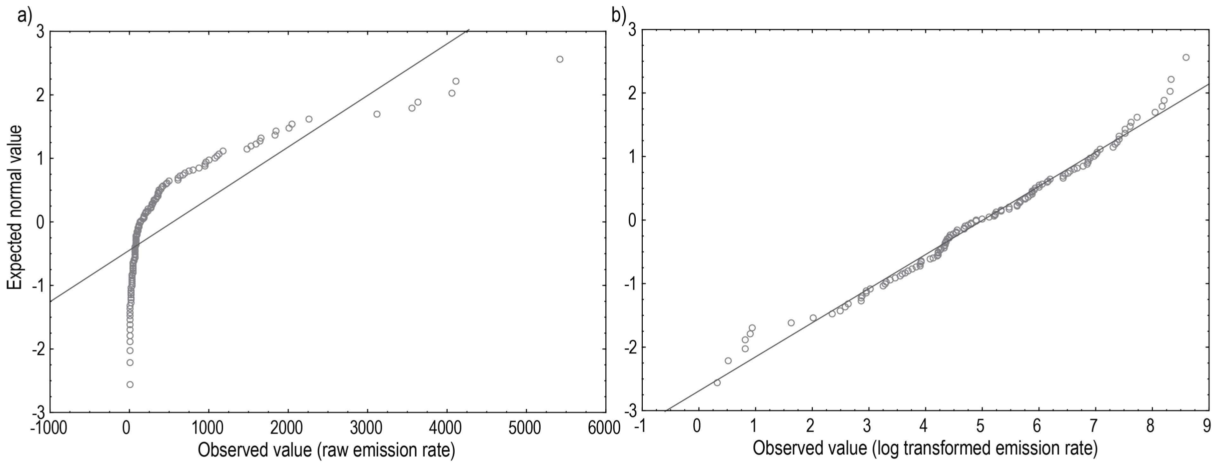


Figure ~~A3~~A4. Normal probability plots for a) raw methane emissions and b) log transformed emissions data. Shapiro-Wilks tests p-value for raw emissions data was < 0.001 and failed the normality test; p-value for log transformed emissions data was 0.081 indicating data was normally distributed.

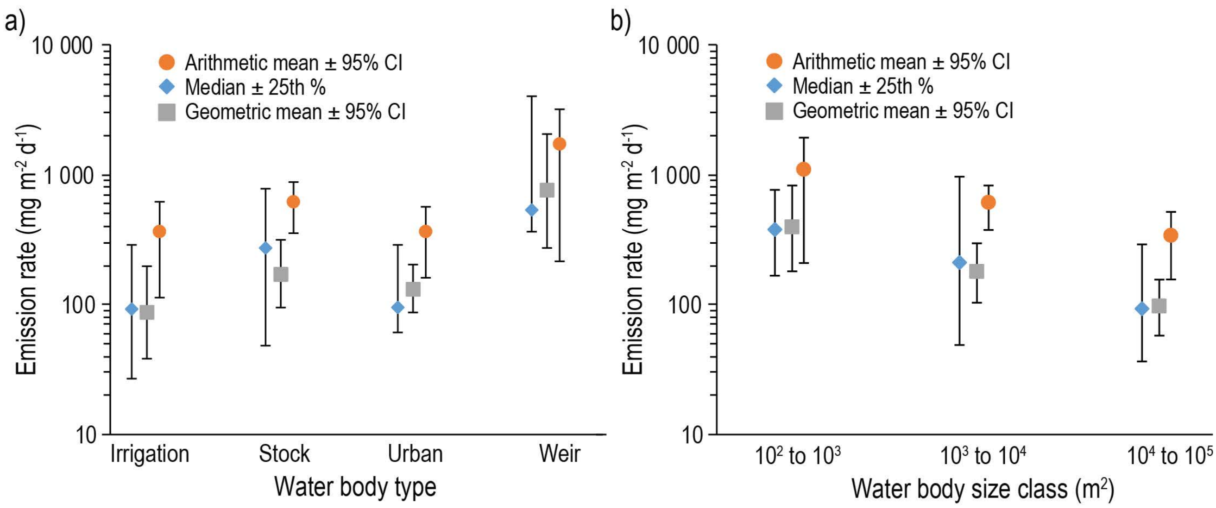


Figure ~~A4~~A6. Three measures of centrality for methane emissions across a) four categories of small water bodies (irrigation dams, stock dams, urban lakes and weirs) and b) three GRanD water body size classes. Error for each measure are as follows: median emission rates and interquartile range (± 25th %), arithmetic and geometric mean emission rates and 95% confidence intervals (± 95% CI).