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Greenhouse gas emissions from waste stabilisation ponds in Western Australia and Quebec (Canada)

Patricia Glaz [a, c, *], Maciej Bartosiewicz [a, b, †], Isabelle Laurion [a, b], Elke S. Reichwaldt [c], Roxane Maranger [b, d], Anas Ghadouani [c]

[a] Institut national de la recherche scientifique, Centre Eau Terre Environnement, 490 rue de la Couronne, Quebec, QC G1K 9A9, Canada
[b] Groupe de Recherche Interuniversitaire en Limnologie et en environnement aquatique (GRIL), Université de Montréal, Montréal, QC H3C 3J7, Canada
[c] School of Civil, Environmental and Mining Engineering, The University of Western Australia, 35 Stirling Highway M015, Crawley, WA 6009, Australia
[d] Département des Sciences Biologiques, Université de Montréal, Montréal, QC H2V 2S9, Canada

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ABSTRACT

Waste stabilisation ponds (WSPs) are highly enriched environments that may emit large quantities of greenhouse gases (GHG), including CO_2, CH_4 and N_2O. However, few studies provide detailed reports on these emissions. In the present study, we investigated GHG emissions from WSPs in Western Australia and Quebec, Canada, and compared emissions to WSPs from other climatic regions and to other types of aquatic ecosystems. Surface water GHG concentrations were related to phytoplankton biomass and nutrients. The CO_2 was either emitted or absorbed by WSPs, largely as a function of phytoplankton dynamics and strong stratification in these shallow systems, whereas efflux of CH_4 and N_2O to the atmosphere was always observed albeit with highly variable emission rates, dependent on treatment phase and time of the day. The total global warming potential index (GWP index, calculated as CO_2 equivalent) of emitted GHG from WSPs in Western Australia averaged 12.8 mmol m^{-2} d^{-1} (median), with CO_2, CH_4 and N_2O respectively contributing 0%, 96.7% and 3.3% of the total emissions, while in Quebec WSPs this index was 194 mmol m^{-2} d^{-1}, with a relative contribution of 93.8, 3.0 and 3.2% respectively. The CO_2 fluxes from WSPs were of the same order of magnitude as those reported in hydroelectric reservoirs and constructed wetlands in tropical climates, whereas CH_4 fluxes were considerably higher compared to other aquatic ecosystems. N_2O fluxes were in the same range of values reported for WSPs in subtropical climate.

1. Introduction

Waste stabilisation ponds (WSPs) have been designed to exploit physical, chemical and biological processes for the treatment of wastewaters. These low-cost, low-energy and low-maintenance systems are widely used in developed and developing countries (Pearson, 1996). Typically, WSPs are shallow water bodies that consist of a series of facultative, maturation and storage ponds, often with two ponds occurring in parallel to increase treatment efficiency. Facultative ponds, receiving raw wastewater, are designed to retain suspended solids and reduce the biological oxygen demand (BOD). Subsequently, maturation ponds are designed to remove pathogens and excessive nutrients, and to lower the BOD (Metcalf and Eddy, 2004). The final treatment phase consists of storage ponds that retain treated water before it is released into the environment or evaporates.

Inland waters have been identified as globally important systems for carbon and nutrient processing (Cole et al., 2007) and have thus been receiving increasing attention in terms of their role in greenhouse gas (GHG) budgets (Bastviken et al., 2011; Wik et al., 2016). Municipal wastewater treatment plants (WWTPs) have also been recognized as potentially high emitters of GHG resulting from high and regular supply of organic matter and nutrients (Daelman et al., 2012). However, relative to natural aquatic ecosystems, comparative and integrated studies on GHG dynamics in WSPs remain scarce. As the importance of WSPs in the global GHG budget is likely to increase worldwide with population growth and the accelerated need for efficient treatment of wastewaters,

* Corresponding author.
E-mail address: glaz.patricia@gmail.com (P. Glaz).
† Present address: Department of Environmental Sciences, University of Basel, Basel, CH-4056, Switzerland.

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comprehensive studies on GHG emissions from such systems are required (Downing et al., 2006).

The amount and composition of biogenic gases emitted from WSPs to the atmosphere can vary diurnally and seasonally depending on specific characteristics of the system. The mixing regime for instance will vary as a function of the incoming rate of water, and together with the overall sewage composition and load, can generate large differences in GHG exchange rates over short and seasonal time scales. The CO_2 evasion to the atmosphere from shallow aquatic ecosystems depends on the ratio of primary production to ecosystem respiration (Kortelainen et al., 2006). Given the typical high phytoplankton biomass in WSPs, CO_2 may be absorbed from the atmosphere through daily photosynthetic uptake (Park et al., 2011). In the case of CH_4, elevated production may be expected given the high organic C loads, elevated decomposition rates and anaerobic conditions of the bottom waters and sediments. Nitrous oxide (N_2O) is produced through nitrification when oxygen is present even at low levels (Mengis et al., 1997), or denitrification under anoxic conditions (Codispotti and Christensen, 1985; Kampschreur et al., 2009), and its emissions may be particularly high in WSPs given the elevated concentration of nutrients combined with oxygen-limited micro-habitats. The responsiveness of CH_4 and N_2O emissions to environmental conditions has received a lot of attention in the last decade (e.g., Blais et al., 2005; Guérin et al., 2008; UNESCO/IHA, 2010), because the warming potential of these gases are 34 and 298 times greater than CO_2 respectively over a 100-year time horizon (IPCC, 2013), but information on emissions of all three gases, and their relative importance from WSPs is rare. GHG emissions may represent an additional environmental cost that needs to be considered when planning and engineering wastewater treatment facilities. In this regard, understanding the dynamics of GHG concentrations and emissions in WSPs will guide future engineering practices.

Considering the growing concern about global climate change and anthropic GHG emissions, a better estimation of the contribution of municipal WSPs to the global GHG budget is needed for future predictions. Furthermore, similarly to small lakes (Downing et al., 2006), the global area of WSPs may be underestimated. In this study, we assess the spatiotemporal dynamics of GHGs in several WSPs in two climatically distinct regions, identify which biological and physicochemical characteristics of wastewaters influence GHG concentrations, estimate GHG diffusive fluxes, and compare our results with previous studies on wastewater treatment and other aquatic ecosystems. The global warming potential of emitted GHG from wastewaters (hereafter the GWP index; given in mmol of CO_2 equivalent m^{-2} d^{-1}) was also calculated as a metric to assess the relative importance of each gas to the overall emissions.

2. Methods

2.1. Study sites and sampling period

Seven WWTPs, for a total of 21 WSPs (average pond length = 75 m; average width = 41 m; depth range = 1.1–2.3 m; area range = 0.08–0.9 ha), were sampled between September 2012 and April 2013 in the southern part of Western Australia (WA). Design and operational variables of the ponds are presented in Table S1. For comparison, two additional WWTPs located close to Quebec City, Canada, in the municipalities of Lac Delage and Stoneham (for an additional 6 WSPs, mean pond length = 75 m; mean pond width = 39 m; mean pond depth = 2.5 m), were sampled between June and October 2010 and in August 2012 (only Lac Delage WWTP in 2012; Table 1). It is worth noting that the amount of ancillary data and the sampling effort was much higher in WA than in Quebec. In particular, three of the seven WWTPs in WA (Boddington, Cranbrook and Wagin) were sampled more intensively, explaining that most results pertain to these, while limnological properties were only available in five of the WWTPs (Table 2). Data also includes discrete and continuous sampling over diurnal cycles in Boddington and Wagin to identify temporal patterns related to physical structure. This part of WA has a Mediterranean-like climate, with mild and wet winters (average air temperature of 13.5 °C and precipitation of 132.5 mm) and hot and dry summers (24.7 °C and 11.4 mm). In Quebec City, summers are relatively warm (average air temperature of 17.7 °C and precipitation of 78.7 mm), but winters are typically cold, windy and snowy (average air temperature of −11.0 °C and total precipitation of 96.5 mm).

2.2. Meteorology, physicochemistry and chlorophyll-a

A Kestrel 4500 meteorological station was installed at Cranbrook, Boddington and Wagin to measure wind speed (±0.1 m s^{-1}), atmospheric pressure, and air temperature, while data were recorded every 30 min as an average over the course of the intervals. Wind speeds were corrected to 10 m height following the law of the wall scaling. Thermistor chains (7 temperature loggers, Onset StowAway TidbiT) were installed at Boddington (maturation pond), Cranbrook (facultative pond) and Wagin (maturation pond) at 20 cm depth intervals from surface to bottom, with temperature (±0.2 °C) recordings every 8 min. Nutrient concentrations (total phosphorus TP, soluble reactive phosphorus SRP, total nitrogen TN, $N-NO_3$ and $N-NH_4$), BOD, coliform counts *(Escherichia coli)*, and total suspended solids (TSS) were assessed once a month at the outlet of the facultative and maturation ponds at Boddington, Cranbrook, Mount Barker, Tambellup and Wagin following standard methods (APHA, 2011). Additionally, at the time of sampling, surface temperature, pH and conductivity were measured using a Hydrolab sensor (TPS WP-81), along with dissolved oxygen (DO) in Boddington, Cranbrook and Wagin with a Handy polaris 2 Oxyguard® sensor. The biomass (as chlorophyll-a, Chl-a) of different phytoplankton groups (cyanobacteria, chlorophytes, diatoms and cryptophytes) was also estimated using a top-bench version of a Fluoroprobe (BBE Moldaenke, Germany) (Beutler et al., 2002) on water transported back to the laboratory in cool and dark conditions (<12 h delay). The Chl-a in Quebec ponds was estimated by spectrophotometry following Nusch (1980).

2.3. Dissolved GHG concentrations

The WSPs in WA were sampled at variable frequency (Table 1), while three of them (Boddington, Cranbrook and Wagin) were sampled at multiple days over an 8-month period to account some of the seasonal variations. To investigate the diurnal variability of GHG concentration, sampling was also performed manually over one 24 h diurnal cycle in March or April 2013 at each of the three main WWTPs, with samples taken every 6 h. In these three WWTPs, discrete sampling was always done close to the inlets and outlets of the ponds, in triplicates. Finally, to further assess the spatial variability of dissolved GHG concentration within a pond, discrete measurements were taken at the inlet, outlet and at two other sites between those in the two facultative ponds of Boddington in March 2015. Three samples were taken at each site for a total of 12 samples per pond within a period of 2 h. The coefficient of variation (CV) was then calculated for each pond. At all times, CO_2 and CH_4 gas samples were collected manually using the headspace method (see below) to quantify surface concentrations (discrete sampling). N_2O was additionally quantified in the three main WWTPs in WA as well as in Quebec (Table 1).

Dissolved GHG concentrations (C_{aq}) were collected using the