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NITROUS OXIDE (N_2O) in MACQUARIE HARBOUR, TASMANIA

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Abstract. Fjord-like estuaries are hotspots of biogeochemical cycling due to steep physicochemical gradients. The spatiotemporal distribution of nitrous oxide (N_2O) within many of these systems is poorly described, especially in the southern hemisphere. The goal of this study is to describe the spatiotemporal distribution of N_2O within a southern hemisphere fjord-like estuary, describe the main environmental drivers of this distribution, the air/sea flux of N_2O, and the main drivers of N_2O production. Cruises were undertaken in Macquarie Harbour, Tasmania to capture N_2O concentrations and water column physicochemical profiles in winter (July 2022), spring (October 2022), summer (February 2023), and autumn (April 2023). N_2O samples were collected at one depth at system end members, and at 5 depths at 4 stations within the harbour.

Results indicate that N_2O is consistently supersaturated (reaching 170% saturation) below the system's freshwater lens where oxygen concentrations are often hypoxic, but infrequently anoxic. In the surface lens, levels of N_2O saturation vary with estimated river flow and with proximity to the system's main freshwater endmember. The linear relationship between AOU and ΔN_2O saturation indicates that nitrification is the process generating N_2O in the system. When river flow was high (July and October 2022), surface water N_2O was undersaturated (as low as 70%) throughout most of the harbour.

When river flow was low (February and April 2023) N_2O was observed to be supersaturated at most stations. Calculated air/sea fluxes of N_2O indicated that the system is generally a source of N_2O to the atmosphere under weak river flow conditions and a sink during strong river flow conditions. The diapycnal flux was a minor contributor to surface water N_2O concentrations, and subhalocline N_2O is intercepted by the riverine surface lens and transported out of the system to the ocean during strong river flow conditions. In a changing climate, Western Tasmania is expected to receive higher winter rainfall and lower summer rainfall which may augment the source and sink dynamics of this system by enhancing the summer / autumn efflux of N_2O to the atmosphere.

This study is the first to report observations of N_2O distribution, generation processes, and estimated diapycnal / surface N_2O fluxes from this system.

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1. Introduction

Despite the fact that fjords and fjord-like estuaries represent only a small portion of the coastal area worldwide they are responsible for sequestering 11% of the global organic carbon (C) burial along terrestrial margins (**Smith *et al.*, 2015**; **Bianchi *et al.*, 2018**, **2020**). These systems are significant sources of greenhouse gasses (GHG) to the atmosphere (**Wilson *et al.*, 2020**; **Rosentreter *et al.*, 2023; Bange *et al.*, 2024**). Many are heavily stratified with strong water column physicochemical gradients (**Acuña-González *et al.*, 2006; Inall and Gillibrand, 2010; Hartstein *et al.* 2019; Salamena *et al.*, 2021, 2022; Maxey *et al.* 2022**). These gradients can be influenced by mesoscale climate drivers like NAO and SAM (see **Austin and Inall 2002; Gillibrand *et al.*, 2005; Maxey *et al.*, 2022**) and local scale drivers like fresh water input and marine intrusions (**Inall and Gillibrand 2010; Hartstein *et al.*, 2019; Maxey *et al.*, 2020; Salamena *et al.*, 2022**).

Nitrous oxide (N_2O) is a potent GHG whose increased presence in the atmosphere is primarily driven by emissions from agricultural soils with an increased presence in poorly oxygenated marine systems (**Laffoley and Baxter 2019**; **Ji *et al.*, 2020**; **Wilson *et al.*, 2020**; **Wan *et* al., 2022; Orif *et al.*, 2023**). With a global warming potential nearly 300 times that of CO_2 (**Myhre *et al.*, 2013**; **Etminan *et al.*, 2016; Eyring *et al.*, 2021; Forster *et al.*, 2021**) N_2O is a key focus of climate studies especially regarding ozone layer depletion. N_2O is a precursor to NO, and a major ozone depleting substance in the atmosphere (**Nevison and Holland, 1997**; **Ravishankara *et al.*, 2009; Portmann *et al.*, 2012**). N_2O production occurs through the microbially mediated processes of ammonia oxidation, nitrite (NO_2^-) reduction, and nitrate (NO_3^-) reduction (**Kuypers *et al.*, 2018**). N_2O production in marine systems is governed by environmental conditions such as dissolved oxygen (DO) availability, ammonium (NH_4^+) availability, light availability, temperature (*e.g.* **Raes *et al.*, 2016**), pH (*e.g.* **Breider *et al.*, 2019**), and microbial community composition (*e.g.* **Wu *et al.* 2020**).

Estuarine systems have disproportionately high biological productivity relative to other marine systems (**Walinsky *et al.*, 2009; Gilbert *et al.*, 2010; Bianchi *et al.*, 2018, 2020**). This also applies to N_2O dynamics with approx. 33% of marine N_2O emissions coming from estuaries (**Bange *et al.*, 1996; Seitzinger *et al.*, 2000**; **Murry *et al.*, 2015**; **Reading, 2022; Rosentreter *et al.*, 2023**). Estuaries can act as net sinks (**Maher *et al.*, 2016; Wells *et al.*, 2018**) and sources (**De Bie *et al.*, 2002; Zhang *et al.*, 2010; Sánchez-Rodríguez *et al.*, 2022**) of N_2O depending on factors controlling air/sea fluxes, waterbody/atmospheric concentrations (**Wells *et al.*, 2018**; **Bange *et al.* 2019**), land use modification (**Reading *et al.*, 2020; Chen *et al.*, 2022**), and even the presence of microplastics (**Chen *et al*, 2022**). Despite the advancements made thus far, our understanding of marine N_2O distribution and atmospheric emissions needs improvement (**Bange *et al.*, 2024**), especially in southern hemisphere fjord-like systems (**Yevenes *et al.*, 2017**). Much of the current uncertainty lies with a lack of in-situ data describing seasonal N_2O dynamics to constrain global emissions models (**Bange *et al.*, 2019**).

The purpose of this study was (1) to investigate the distribution and seasonal variability of N_2O concentrations and emissions in a southern hemisphere fjord-like estuary and (2) to decipher the major physical and biological drivers of the N_2O emissions.

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2. Methods

2.1 Study Area

Macquarie Harbour is a southern hemisphere fjord-like estuary located on Tasmania's west coast (**Figure 1**). The harbour is oriented NW by SE, and is approximately 33 km long, 9 km wide, with a surface area of 276 km^2. The mouth of the harbour is constricted by a shallow (4-8m), long (14km) sill known as "Hells Gates". Hells Gates muffles tidal forcing resulting in harbour water levels primarily determined by river flow and wind set up (**Hartstein *et al.*, 2019**). The morphology of this system results in sharp gradients of DO, salinity, and temperature which are seasonally dependant (**Creswell *et al.*, 1989**; **Hartstein *et al.*, 2019**; **Maxey *et al.*, 2022**). In surface waters DO concentrations are nearly always in equilibrium with the air but decrease sharply through the halocline (~8m to 15m). Subhalocline layers (~15m to a few meters from the bottom) are observed to be below 62.5 µM more than 50% of the time (*see* **Maxey *et al.*, 2022**). Near the seabed, episodic marine intrusions (deep water renewal) refresh the supply of DO. In the upper reaches of the harbour marine intrusions are much less common (*see* **Hartstein *et al.*, 2019**; **Maxey *et al.*, 2022**). In these areas the DO concentration falls below 31 µM nearly a third of the time (**Maxey *et al.*, 2022**).

Like many fjord-like estuaries, the distribution of DO is driven by multiple physical and biological processes whose relative importance depends on position along the estuarine axis as well as water column depth (**Hartstein *et al.*, 2019** and **Maxey *et al.*, 2017, 2020, 2022**). There is almost no DO produced below the halocline (8m to 12m deep) as the overlying freshwater lens is high in chromophoric dissolved organic matter (CDOM) limiting light available to primary producers in the surface water layers (within 3m) (**Maxey *et al.*, 2017, 2020**). Basin water oxygen concentrations and salinity are largely influenced by advection of marine water over Hells Gates (**Andrewartha and Wild-Allen 2017; Hartstein *et al.*, 2019; Maxey *et al.*, 2022**). This process is driven by low atmospheric pressure, sustained NW winds, and low catchment rainfall which itself is influenced by Southern Annular Mode (SAM) (**Hartstein *et al.*, 2019; Maxey *et al.*, 2022**). Hydrodynamic and oxygen tracer numerical simulations of the harbour by **Andrewartha and Wild-Allen (2017)** support measurements by **Maxey *et al.* (2022)** which show that basin water residence times and basin water oxygen concentrations are tied to the flow conditions of the freshwater endmembers. In their model basin water tracer concentrations were reduced by half in approximately 65 days during low river flow conditions (approximately 70 days at the surface) and in approximately 110 days during normal flow conditions (approximately 40 days at the surface).

The main source of freshwater to the harbour is located on its southeast end (the Gordon River) and drains a nearly pristine catchment (including the Franklin River) of approximately 5,682 km^2 (**Macquarie Harbour Dissolved Oxygen Working Group, 2014**; **Fig 1**). The Gordon River discharges an estimated 180,000 tons organic carbon (OC) per year (**Maxey *et al.*, 2020, 2022**) into the estuary. It should be noted that this area receives the some of the highest rainfall (more than 2,500 mm $year^{-1}$) volume in Australia (**Dey *et al.*, 2019**). The King River, located on the harbour's northern end, is the second largest contributor of fresh water to the estuary and drains a catchment area of 802 km^2. Unlike the Gordon River, the King River has a history of receiving treated mining (*e.g.* copper) effluent and transporting this to the harbour (**Carpenter *et al.*, 1991; Teasdale *et al.*, 2003**).

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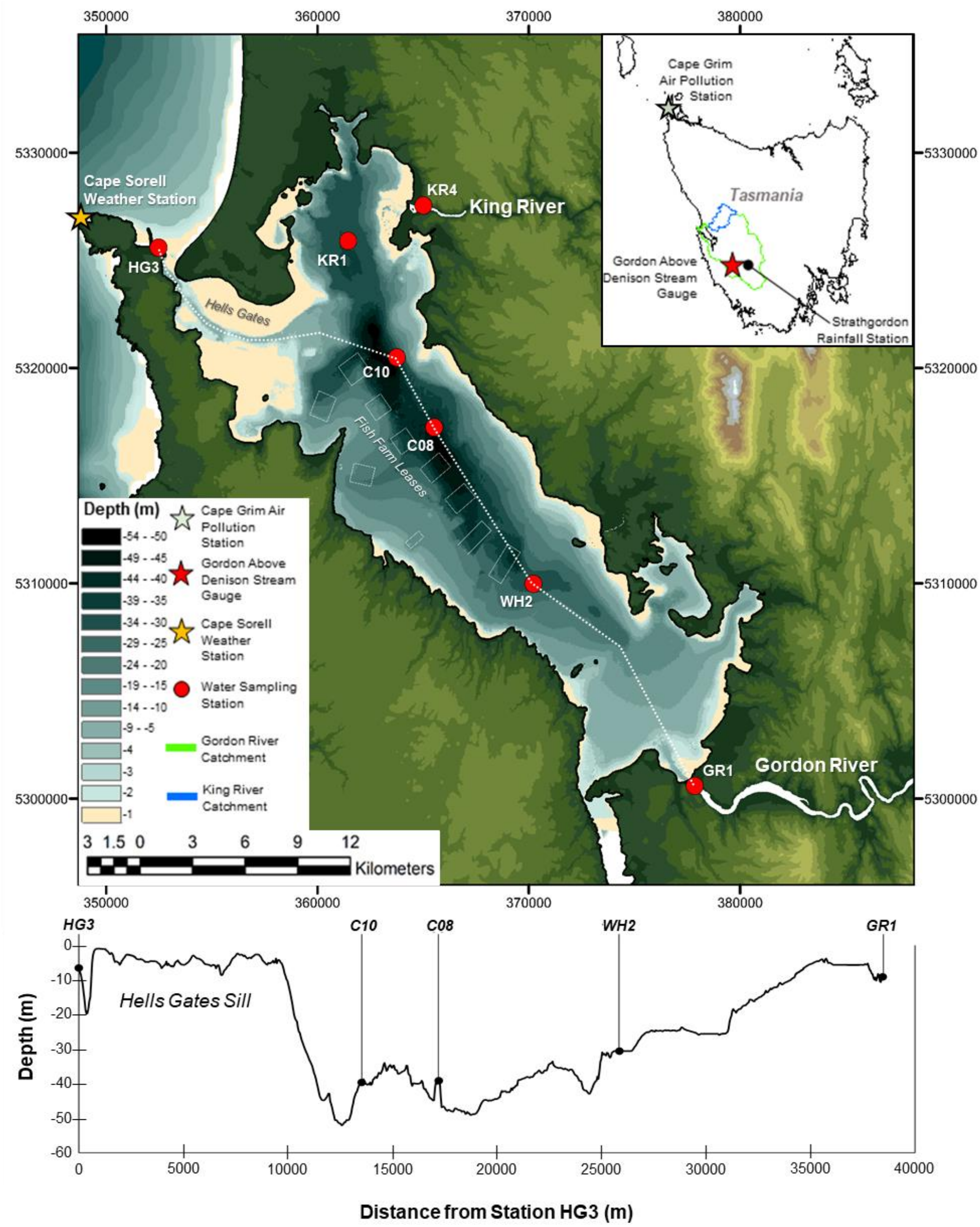

115 **Figure 1: Macquarie Harbour, Tasmania. Water sampling stations shown with red circles; Cape Grim Air Pollution monitoring station shown as a green star (see inset map). Cape Sorell Weather Station shown as an orange star. Gordon Above Denison stream gauge shown as a red star (see inset map). Aquaculture lease boundaries are shown as hollow rectangles. Lease locations are sourced from Land Information Systems Tasmania (LISTmap - https://maps.thelist.tas.gov.au/). Station names reflect general harbour locations where KR1 indicates King River 1;**
120 **C10 and C08 indicate Central Harbour 10 and 08 respectively; WH2 indicates World Heritage Area 2; and GR1 indicates Gordon River station 1. Coordinates are displayed in GDA_1994_MGA_Zone_55. Bathymetry through the system shown as a dashed line, note that this track excludes stations KR4 and KR1.**

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2.2 Experimental Design

N_2O distribution was assessed by collecting water samples across 7 stations, including the harbour's endmembers (mouths of the Gordon and King Rivers as well as the harbour mouth at Hells Gates Inlet; *see* **Figure 1** and **Table 1**) and stations along the longitudinal axis of the harbour where the deepest basins are located (named KR1, C10, C08, and WH2). Samples collected at endmember stations were collected from a single depth as these stations are shallow. Samples in the harbour body were collected at 5 depths from the surface (2m) to approx. 1m from the seabed. Collection campaigns were conducted in July 2022, October 2022, February 2023, and April 2023. At each station and depth three replicate vials (n = 3) were collected for the determination of N_2O concentration.

2.3 Field Sampling

At each station, water quality sonde profiles were collected from the surface to the seabed at 1 meter intervals using a YSI EXO-1 equipped with optical DO (accuracy from 0 to 625 µM ± 3 µM or 1% of reading whichever is greater; precision is 0.03 µM), salinity (accuracy ± 0.1 or 1% of reading whichever is greater; precision is 0.01), temperature (accuracy is ± 0.15 ºC; precision is 0.01 ºC), and depth sensors. Sonde calibration was checked and corrected (when needed) each sampling period.

Water samples were collected at various depths (see **Table 1**) using a 5 L Niskin bottle sampler. Water sample parameters include dissolved Total Ammoniacal N (NH_3 + NH_4^+) (TAN), NO_3^-, and N_2O. N_2O samples were collected in triplicate immediately after retrieval of the Niskin bottle by transferring water from the bottle through silicone tubing into a 20 mL borosilicate vial. Sample water was added to the vial by placing the tubing at the bottom and allowing the vial to overflow several volumes before sealing with a butyl rubber stopper and aluminium crimp. After ensuring the sample vial is bubble free, 50 µL of saturated mercury chloride ($HgCl_2$) solution was injected into the sample to arrest biological activity. All N_2O samples were shipped to GEOMAR in Kiel, Germany for analysis. Samples were measured in July/August 2023 within 4 to 12 months after sampling and were not affected by the storage time (**Wilson *et al.*, 2018**).

Water collected for dissolved inorganic nitrogen (N) was filtered immediately using 0.45 µm polyethersulfone syringe filters (Whatman Puradisc). Samples were stored in a chilled dark container until being transported to Analytical Services Tasmania in Hobart, Australia for analysis. Dissolved Total TAN and NO_3^- were analysed using a Lachat Flow Injection Analyser. TAN and NO_3^- analyses used methods based on APHA Standard methods (2005) 4500-NH_3 H (reporting limit 0.005 mg L^{-1}) and 4500 - NO_3^- L^{-1} (reporting limit 0.002 mg L^{-1}).

Table 1: Sampling stations showing coordinates, parameters, and sampling depth (in meters).

Station	Station Depth (m) (MSL)	Dissolved Oxygen Salinity Temperature	N_2O	TAN (NH_3 + NH_4^+)	NO_3^-
HG3 *352484, 5325594*	8	*Every Meter*	*5m*	*5m*	*5m*

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KR4 *365018, 5327550*	3	*1m*	*1m*	*1m*	*1m*
KR1 *361316, 5325972*	*36*	*Every Meter*	*2, 12, 20, 30, 35m*	*2, 12, 20, 30, 35m*	*2, 12, 20, 30, 35m*
C10 *363708, 5320464*	*44*	*Every Meter*	*2, 12, 20, 30, 42m*	*2, 12, 20, 30, 42m*	*2, 12, 20, 30, 42m*
C08 *365489, 5317238*	*47*	*Every Meter*	*2, 15, 25, 35, 45m*	*2, 15, 25, 35, 45m*	*2, 15, 25, 35, 45m*
WH2 *370218, 5309894*	*32*	*Every Meter*	*2, 12, 20, 25, 30m*	*2, 12, 20, 25, 30m*	*2, 12, 20, 25, 30m*
GR1 *377784, 5300603*	*12*	*Every Meter*	*10m*	*10m*	*10m*

2.4 Analysis of Rainfall and River Loading Estimation

River loading and rainfall were analysed using methods presented in **Maxey *et al.* (2022)** where rainfall and stream gauge data were collected from the Gordon River catchment, Strathgordon rainfall gauge station and the Gordon Above Denison (GAD) stream gauge (**Figure 1**). The rainfall and flow metrics computed include the average daily rainfall over a 20-day period prior to sampling; total accumulated rainfall 20,10, 5, and 3 days prior to sampling; estimated Gordon River flow into the estuary; and measured flow at the GAD stream gauge.

Gordon River flow was estimated by scaling daily rainfall to the size of the catchment and assuming a rainfall and runoff coefficient of 0.70 adopted from a neighbouring catchment with similar land cover, geology, and slope (**Willis, 2008**). Additional streamflow from Gordon River dam releases was estimated by subtracting scaled rainfall contributions to river flow measured at the GAD stream gauge. This flow was added to the estimated runoff entering the harbour. Rainfall and flow data were provided by the Australian Bureau of Meteorology (BOM).

NO_3^- and TAN loading was estimated my multiplying the measured concentration of each parameter at station GR1 (*see* **Figure 1** and **Table 1**) by the estimated Gordon River flow.

2.5 Analysis of Water Column N_2O Concentrations, Air/Sea Flux, and Diapycnal Flux

2.5.1 Determination of N_2O Concentrations

Water samples were analysed for N_2O using the static-headspace equilibration method followed by gas chromatographic separation (HP Agilent 5890) and detection with an electron capture detector (ECD) as described in **Bange *et al,* (2019)**, **Bastian (2017)**, and **Kallert (2017)**. The concentrations of N_2O in the samples was calculated with the following equation (**Equation 1;** *see* **Bange *et al.*, 2006**):

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Equation 1

$$C_{obs} = \frac{x' P V_{hs}}{RTPV_{wp}} + X'\beta P$$

C_{obs} is the concentration (nmol L^{-1}) of N_2O in the water sample; **x'** is the measured dry mole fraction of N_2O in the sample vial's headspace; *P* is the ambient pressure set to 1 atm; ***V_{hs}*** and ***V_{wp}*** are the volumes of the headspace in the vial and water in the vial; ***R*** is the gas constant; ***T*** is the temperature during equilibrium; and ***β*** is the solubility of N_2O (**Weiss and Price, 1980**). The mean relative error of the concentration values obtained was 2.4% (± 0.16).

2.5.2 Estimation of N_2O Air/Sea Fluxes and N_2O Saturations

N_2O air/sea fluxes (***F*** in µmol m^{-2} d^{-1}) were estimated using equations from **Zhang *et al.*, (2010)** and **Bange *et al.*, (2019)** (**Equation 2)** *Where*:

Equation 2

$$F = K * (C_{obs} - C_{eq})$$

C_{obs} is the measured concentration (nmol L^{-1}) of N_2O in the water sample; ***C_{eq}*** is the air-equilibrated seawater N_2O concentration, calculated for in situ temperature and salinity using the solubility data of **Weiss and Price (1980)**. ***K*** is the gas transfer velocity, which in the absence of direct measurements can be expressed as a function of the wind speed and the Schmidt Number (*Sc*). For this study we sourced daily average wind speed from the Cape Sorrel Weather Station at the northern end of Macquarie Harbour (http://www.bom.gov.au/climate/data/index.shtml station ID 097000; see **Figure 1** for station location). ***K*** was estimated using relationships in **Raymond and Cole** (**2001**). Fluxes at Macquarie Harbour's endmember stations used ***K*** values that account for additional forcings like bottom sheer (*see* **Raymond and Cole 2001**; **Zappa *et al.*, 2003**; **Abril and Borges 2004, Beaulieu *et al.*, 2012; Rosentreter *et al.*, 2021**). Deeper stations in the harbour's main body (*i.e.* KR1, C10, C08, WH2) have surface layers which are separated from the seabed by more than 10 meters. A wind-based K_{600} estimator was used to estimate air-sea flux in those locations (*see* **Raymond and Cole 2001).** Atmospheric N_2O for this estimation was sourced from monthly mean baseline greenhouse gas mole fractions measured at the Kennaook / Cape Grim Baseline Air Pollution Station, located in north west Tasmania. This station measures atmospheric N_2O using a gas chromatograph (GC) equipped with an ECD (https://www.csiro.au/en/research/natural-environment/atmosphere/latest-greenhouse-gas-data). N_2O saturation (in %) were computed as N_2O saturation = 100 * C_{obs} *C_{eq}^{-1}.

2.5.3 Estimation of Diapycnal N_2O Flux

N_2O diapycnal fluxes (***F_{dia}*** **; Equation 3**) from basin waters (sample depths of 20m or 25m) to the harbour's surface lens (sample depths of 2m) were estimated as:

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Equation 3

$$\mathbf{F_{dia} = K\rho \frac{d[N_2O]}{dz}}$$

Where ***z*** is depth. Diapycnal diffusivity (***K_ρ*; Equation 4**) was computed with the local buoyancy frequency (N^2), ***Γ*** set to 0.2 (Osborn 1980), and **ε** the dissipation rate of turbulent kinetic energy assumed to be on the upper end of values for the mixing zone of stratified systems 1 x 10^{-5} (**Arneborg *et al.*, 2004; Mickett *et al.*, 2004; Fer *et al.*, 2006**).

Equation 4

$$\mathbf{K_\rho = \Gamma \frac{\varepsilon}{N^2}}$$

2.6 Data Analysis

The relationships between N_2O saturation and water quality parameters such as DO concentration, salinity, temperature, nitrate, and ammonium concentrations were analysed using Pearson correlation. The effects of season and depth on N_2O saturation at each sampling station was tested using a 2-way ANOVA. The relationship between rainfall / river flow metrics from the Gordon River and surface water N_2O saturation / N_2O air/sea flux at each station was analysed using Pearson correlation.

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3. Results

3.1 Rainfall and River Loading

Twenty-day rainfall accumulation ranged from a low of 117 mm in July 2022 to a high of 139 mm in April 2023 with no detectable seasonal differences (*see* **Figure 2A**). Average daily rainfall was similar across all months and ranged from 5.12 (± 2.57) mm in July 2022 to 5.79 (± 3.03) mm in October 2022 (*see* **Figure 2B**). As observed with accumulation metrics no seasonal differences were detected.

Estimated flow at the Gordon River mouth and GAD stream gauge were greater in July and October 2022 than February and April 2023 (**Figure 2C**). At the GAD stream gauge, average flows were observed to decrease during the study period. Greatest flow was observed in winter (July 2022) at 107.6 (± 15.9) $m^3\ s^{-1}$ and lowest in autumn (April 2023) at 30.5 (± 2.2) $m^3\ s^{-1}$ (**Figure 2D**).

Estimated NO_3^- and TAN loading varied with NO_3^- loads of 1.69 tonnes day^{-1} observed in July 2022, dipping to 0.31 tonnes day^{-1} in October 2022, and then rising again to 1.77 and 2.77 tonnes day^{-1} in February and April 2023 (**Figure 2E**). TAN loading mirrored this pattern with peaks occurring in October 2022 and February 2023 and lows occurring in July 2022 and April 2023. Patterns in N_2O loading from the Gordon River were similar to those observed for NO_3^- (**Figure 2F**).

3.2 Water Column Physicochemical Profiles

DO profiles at the stations located within the main body of the harbour show a well oxygenated surface layer that rapidly attenuates with depth (**Figure 3A**) through the halocline (**Figure 3B**). There is a prominent riverine surface lens in the main harbour extending to depths of up to 8m depending on sampling period and location within the estuary. Salinity in the surface waters was lower in July and October 2022 (6 to 13) than February and April 2023 (greater than 20). Below the halocline salinity ranged from approx. 28 to 32.

The DO gradient between the surface and subhalocline waters was steeper in October relative to July 2022 with October 2022 DO concentrations approaching single digits (3.1 µM) at station WH2, nearest the Gordon River mouth (*see* **Figure 1**). In general, the subhalocline concentrations of DO were lower with proximity to the Gordon River mouth. The temperature of the freshwater surface layer ranged from about 9 ºC to 19 ºC, but showed little variation below the halocline where temperature ranged between 13 ºC to 16 ºC (**Figure 3C**).

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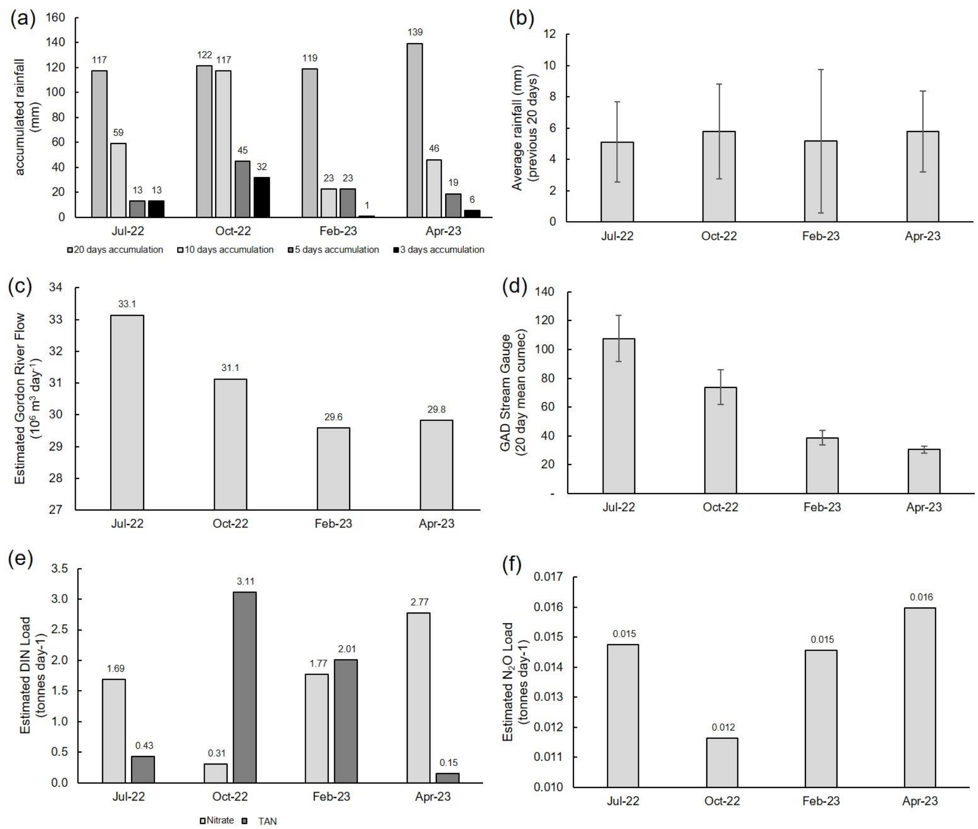

265 **Figure 2: Rainfall and estimated Gordon River loading estimates for each sampling event. A) accumulated rainfall (mm) 10, 5, and 3 days prior to each sampling event; B) average (mean) daily rainfall over a 20 day period prior to each sampling event; C) estimated Gordon River Flow into the harbour in millions of m^3 day^{-1}; D) daily mean flow (m^3 sec^{-1}) over previous 20 days prior to sampling (± standard error) at the Gordon Above Denison Stream Gauge; E) estimated nitrate and ammonium loads entering the harbour from the Gordon River; F) estimated N_2O load (tonnes day^{-1}) entering the harbour from the Gordon River.**

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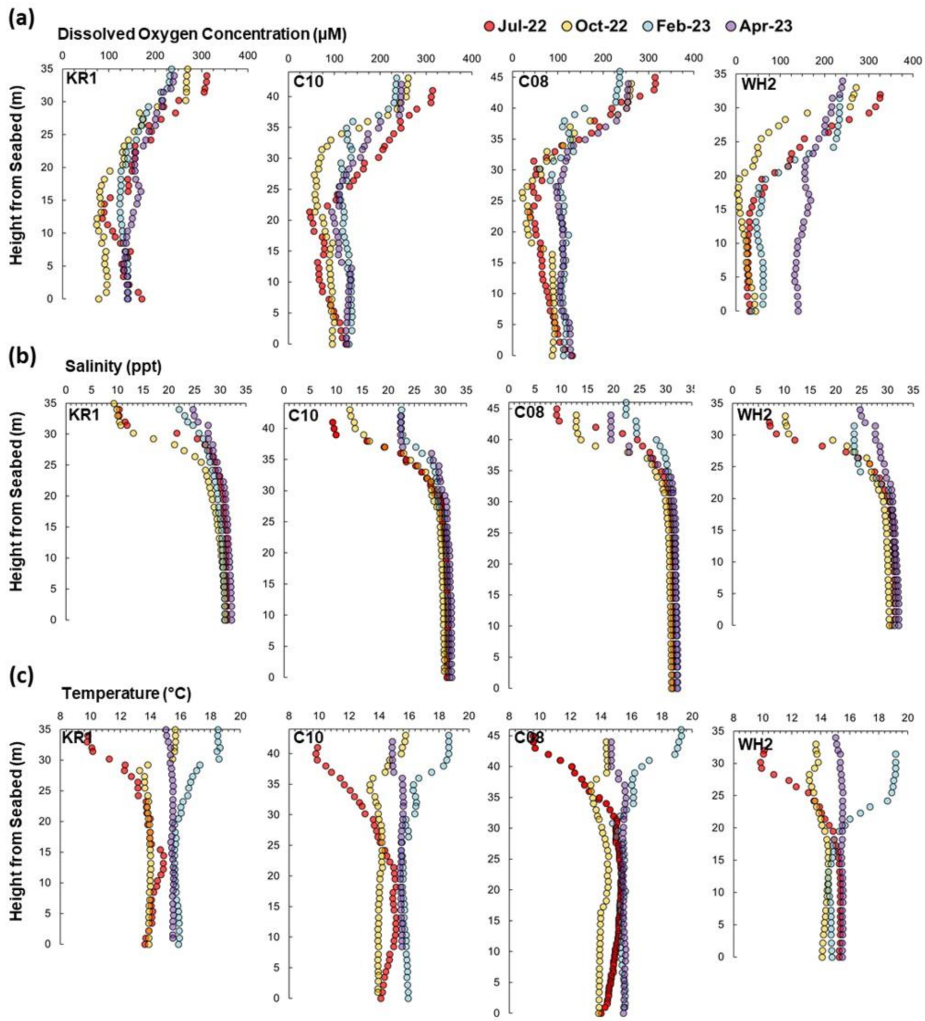

Figure 3: Dissolved oxygen (μM) (Row A), salinity (Row B), and temperature (°C) (Row C) profiles (referencing height from seabed) collected at stations KR1, C10, C08, and WH2 in July 2022 (red dots), October 2022 (yellow dots), February 2023 (blue dots), and April 2023 (purple dots). Measurements were made every 1 meter.

3.3 N_2O Distribution

At each harbour station, depth and season (and their interaction) significantly impacted N_2O saturation (two-way ANOVA, $\alpha = 0.05$, *degree of freedom (d.f.)* = 59). At 2 m, N_2O saturation was observed to be below 100% at all stations in July 2022 (**Figure 4**) and at stations KR1, C10, and C08 in October 2022. In February and April 2023 N_2O saturation in the harbour was above 100% through the water column except in KR1 surface waters. The maximum N_2O concentrations were observed in the subhalocline. Among the subhalocline observations the maximum N_2O concentrations (reaching over 170%) were observed at the base of the Hells Gates sill at station C10 in October 2022.

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All endmember N_2O concentrations were undersaturated in July 2022. In October, stations KR1 and HG3 were observed to be approx. 100% saturated but N_2O at station GR1 was undersaturated. In February and April 2023 N_2O concentrations were supersaturated at all endmember stations. There were statistically significant linear correlations between N_2O saturation and salinity ($r = 0.494$; $p = 5.5 \times 10^{-7}$, $n = 92$), temperature ($r = 0.391$; $p = 1.2 \times 10^{-4}$, *d.f.* = 90), DO concentration ($r = -0.563$; $p = 5.2 \times 10^{-9}$, *d.f.* = 90), and nitrate concentration ($r = 0.559$; $p = 6.9 \times 10^{-9}$, *d.f.* = 90) in the harbour stations (**Figure 5**). The correlation between N_2O saturation and the TAN concentration however was not statistically significant ($r = 0.174$; $p = 0.31$, *d.f.* = 34).

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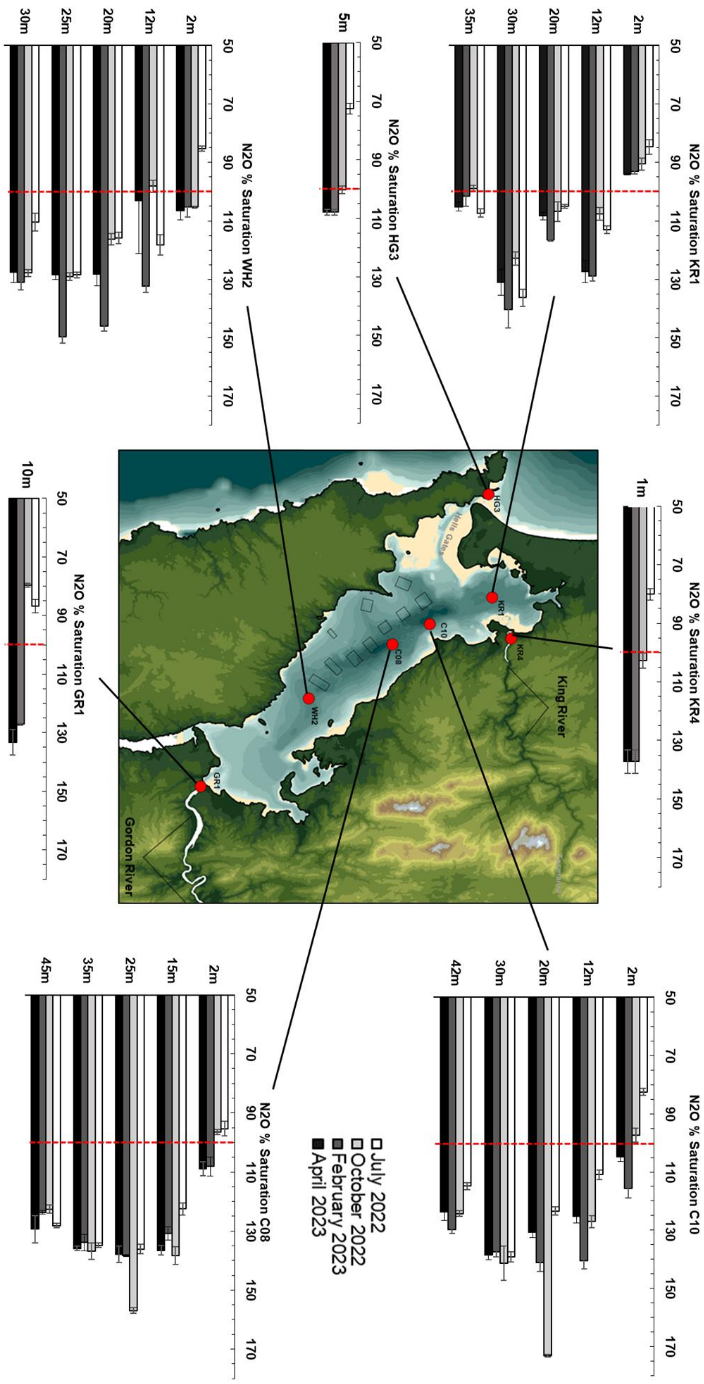

Figure 4: Mean (± standard error) N_2O % saturation observed at each sampling station, with depth, and across seasons. Note that a red dashed line indicating 100% at the time of sampling has been placed on each panel for reference.

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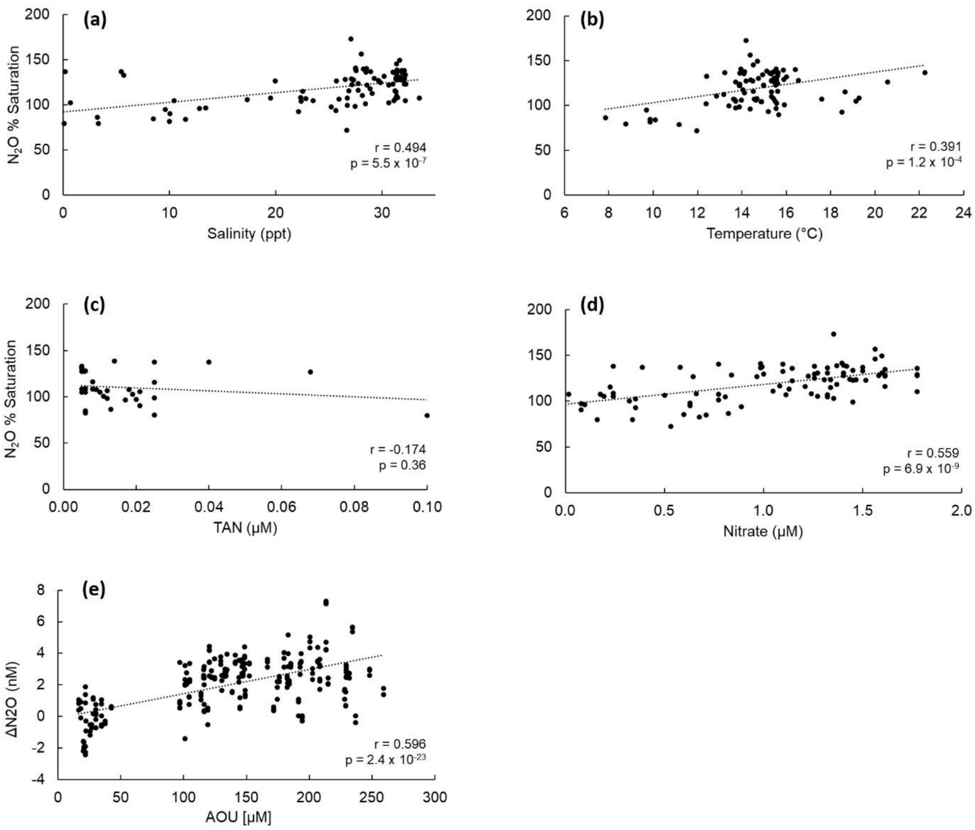

Figure 5: Correlation between N_2O % Saturation observed across the harbour and A) Salinity, B) Temperature, C) Total Ammoniacal Nitrogen (TAN) concentration, D) Nitrate concentration. The correlation between AOU [μM] and ΔN_2O [nM] is shown in panel E. Pearson correlation coefficients (r) and their associated p value are shown in each panel.

3.4 N_2O Air/Sea and Diapycnal Fluxes

Atmospheric N_2O mole fractions measured at Kinnaook / Cape Grim Air Pollution Station (*see* **Figure 1**) were observed to increase from 334.7 ppb in July 2022 to 335.9 ppb in February 2023 (*see* Error! Reference source not found.). The April 2023 atmospheric N_2O mole fraction was slightly lower than that observed in February 2023 at 335.6 ppb.

Estimated N_2O air/sea flux in the main harbour stations (KR1, C10, C08, WH2) was observed to range from -12.88 (± 0.88) μmol N_2O m^{-2} day^{-1} at C10 in July 2022 (negative sign indicates absorption of N_2O into the surface waters from the atmosphere) to 7.31 (± 0.88) μmol N_2O m^{-2} day^{-1} at the same station in February 2023 (using the "High" K_{600} estimator from **Raymond and Cole (2001)**; *see* **Table 2**).

Station KR1 was always observed to be a net sink for atmospheric N_2O, and every non-endmember station was an estimated sink in July 2022. Near the head of the system, station WH2 was observed to be a net source of

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N_2O to the atmosphere in October 2022, February 2023, and April 2023, as were stations C10 and C08 (positioned above the deepest basins) in February 2023 and April 2023.

Estimated diapycnal fluxes using local buoyancy frequencies showed a consistent upwards movement of N_2O from the subhalocline to surface layers with the smallest fluxes observed in July 2022 (49 nmol N_2O m^{-2} day^{-1} at C08) and largest fluxes observed in October 2022 (up to 1308 nmol N_2O m^{-2} day^{-1} at WH2) and February 2023 (up to 1200 nmol N_2O m^{-2} day^{-1} at C10) (**Table 3**). Patterns in the size of the diapycnal flux generally reflected the patterns of N_2O % saturation with the largest fluxes occurring in October 2022 during the periods of greatest N_2O % saturation. Overall the magnitude of the estimated diapycnal fluxes was smaller than estimated air/sea fluxes (smaller) however in February the fluxes were of similar magnitudes.

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Table 2: Estimated sea-to-air N_2O flux (mean µMol N_2O m^{-2} day^{-1} ± standard error) of the main harbour stations using calculations presented in Bange et al. (2019) and Zhang et al. (2020) and a range of k_{600} estimators from Raymond and Cole (2001). Low, Mid, and High represent different estimators of k_{600} presented in Raymond and Cole (2001). Positive values indicate the flux of N_2O from the harbour water to the atmosphere. Negative values (shown in with bold text) indicate flux of N_2O from the atmosphere into the harbour water. Estimated Gordon River Flow and Mean (20 day) Gordon Above Dennison (GAD)Stream Gauge are also shown for each month as well as the Pearson Correlation and associated p-values between flow metrics, rainfall, and sea-to-air flux (and surface water % saturation).

Station	K_{600} Est.	Jul 2022 µmol N_2O m^{-2} day^{-1}	Oct 2022 µmol N_2O m^{-2} day^{-1}	Feb 2023 µmol N_2O m^{-2} day^{-1}	Apr 2023 µmol N_2O m^{-2} day^{-1}	Gordon Flow vs Surface Flux	GAD Flow vs Surface Flux	GAD Flow vs % N_2O Sat.	Rainfall vs Surface Flux
KR1	High:	**-11.07 ± 1.84**	**-04.01 ± 0.86**	**-03.30 ± 0.49**	**-03.17 ± 0.16**				
	Mid:	**-08.45 ± 1.41**	**-03.19 ± 0.69**	**-02.55 ± 0.38**	**-02.44 ± 0.12**	r = -0.8316 p = 7.5 x 10^{-4}	r = -0.8624 p = 3.1 x 10^{-4}	r = -0.8726 p = 2.1 x 10^{-4}	r = 0.5577 p = 0.060
	Low:	**-04.69 ± 0.78**	**-01.93 ± 0.42**	**-01.46 ± 0.22**	**-01.38 ± 0.07**				
C10	High:	**-12.88 ± 0.88**	**-01.21 ± 1.07**	07.31 ± 1.57	02.60 ± 0.85				
	Mid:	**-09.83 ± 0.67**	**-00.96 ± 0.85**	05.65 ± 1.22	02.00 ± 0.66	r = -0.8298 p = 8.4 x 10^{-4}	r = -0.9091 p = 4.2 x 10^{-5}	r = -0.8795 p = 1.6 x 10^{-4}	r = 0.2751 p = 0.387
	Low:	**-05.46 ± 0.37**	**-00.58 ± 0.51**	03.22 ± 0.69	01.13 ± 0.37				
C08	High:	**-03.50 ± 1.82**	**-01.69 ± 0.38**	04.08 ± 1.07	04.57 ± 1.79				
	Mid:	**-02.67 ± 1.39**	**-01.34 ± 0.29**	03.15 ± 0.83	03.52 ± 1.38	r = -0.8547 p = 3.97 x 10^{-4}	r = -0.8804 p = 1.6 x 10^{-4}	r = -0.8447 p = 5.4 x 10^{-4}	r = 0.1846 p = 0.566
	Low:	**-01.49 ± 0.77**	**- 0.81 ± 0.18**	01.80 ± 0.47	01.98 ± 0.78				
WH2	High:	**-10.88 ± 0.68**	02.63 ± 0.17	02.40 ± 1.56	03.50 ± 1.72				
	Mid:	**-08.30 ± 0.52**	02.09 ± 0.14	01.85 ± 1.20	02.69 ± 1.33	r = -0.8071 p = 1.51 x 10^{-3}	r = -0.8269 p = 9.1 x 10^{-4}	r = -0.8077 p = 1.5 x 10^{-3}	r = 0.6316 p = 0.028
	Low:	**-04.61 ± 0.29**	01.26 ± 0.08	01.06 ± 0.69	01.52 ± 0.75				
Gordon River Flow (m^3 sec^{-1})		383.6 ± 38.9	360.3 ± 54.1	342.6 ± 74.6	324.3 ± 26.6	-	-	-	-
GAD Flow (m^3 sec^{-1})		107.6 ± 15.9	73.7 ± 12.1	38.8 ± 5.1	30.5 ± 2.2	-	-	-	-

Table 3: Estimated diapycnal N_2O flux (nmol N_2O m^{-2} day^{-1}) from 20 m to 2 m within the main harbour stations Positive values indicate the flux of N_2O from the basin water (20 m) to the surface lens (2m).

Station	July 2022 nmol N_2O m^{-2} day^{-1}	October 2022 nmol N_2O m^{-2} day^{-1}	February 2023 nmol N_2O m^{-2} day^{-1}	April 2023 nmol N_2O m^{-2} day^{-1}
KR1	80	282	992	395
C10	140	1200	1040	454
C08	49	782	778	348
WH2	117	125	1308	240

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4. Discussion

Our study is the first to report on N_2O distribution and air/sea flux from an Australasian fjord-like estuary. We set out to investigate how N_2O concentrations varied along horizontal and depth gradients; how N_2O concentrations and estimated surface water emissions vary seasonally; how N_2O concentrations vary with freshwater inputs; and whether the relationship between AOU and ΔN_2O could help clarify the primary mechanism for N_2O generation in this system.

We used surface water observations, local wind speed (from Cape Sorell weather station) and atmospheric N_2O mole fractions (from Cape Grimm; **Figure 1**) to estimate N_2O air/sea flux (based on **Zhang *et al.*, (2010)** and **Bange *et al.*, (2019)**) and found that Macquarie Harbour functions as both a sink and a source of N_2O. Most harbour stations were estimated to be a sink for N_2O in July and October 2022 (when river flow was greater) and a source in February and April 2023 (during low river flow periods; *see* **Figure 6** and **Table 2**). Pearson correlations show that when freshwater flow is high N_2O air/sea flux is negative (indicating uptake from the atmosphere) and when freshwater flow is low N_2O air/sea flux is positive (**Table 2**). Our observations highlight that freshwater flow is a key driver of N_2O emissions in this estuary. In addition, Gordon River flow is heavily influenced by hydroelectric dam release (up to ~28% of the flow in July 2023). Rainfall in the catchment area may offset the effects of dam release, but our observations did not capture this as rainfall itself was not significantly correlated with N_2O concentrations or air/sea flux.

The river endmember concentrations of N_2O were often observed to be undersaturated, as observed in the South Platte River Basin, USA, **McMahon and Dennehy 1999**; Neuse River Estuary, USA, **Stow *et al.*, 2005**; headwater streams, Ontario, Canada, **Baulch *et al.*, 2011**; Upper Mara River Basin, Kenya, **Mwanke *et al.*, 2019.** Our observations of endmember N_2O concentrations were similar to the lower end of the concentrations reported in **McMahon and Dennehy (1999)** (approx. 80% saturation), but not as low as those reported Jackson Creek, Ontario, Canada in **Baulch *et al.*, (2011)** were some observations reached <20% saturation. N_2O undersaturation in those systems was attributed to complete denitrification (use of N_2O as a terminal electron acceptor by denitrifies) in streams with high DOC loads, low DO, low NO_3^- concentrations. It should also be noted that up to 28% of the estimated Gordon River flow was found to be associated with flow through the Gordon Above Dennison stream gauge (a proxy for hydroelectric dam/reservoir release to the Gordon River). Boreal reservoirs have been shown to be net sinks of atmospheric N_2O (**Hendzel *et al.*, 2005**) which was attributed to increased N_2O demand to drive complete denitrification. There is good reason to believe that N_2O may be scavenged in the Gordon and King Rivers as well because they do often have high DOC concentrations, high water column DO demand (**Maxey *et al.*, 2020**), and low DO concentrations in near the stream bed (**Maxey *et al.*, 2022**).

Below the estuary's predominately freshwater surface lens, the fjord-like morphology drives suboxic conditions like those observed in the subhalocline waters at station WH2 in October 2022 (*see* **Figure 3; Hartstein *et al.*, 2019; Maxey *et al.*, 2020, 2022**). While these conditions do not always persist, DO concentrations below 31 µM have been observed to occur more than 30% of the time up estuary, specifically at station WH2 (**Maxey *et al.*, 2022**). In layers of the harbour where DO concentrations were lowest (subhalocline layers) we observed the

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maximum N_2O concentrations (**Figure 4**). Subhalocline N_2O saturation was observed to generally range from approx. 110% to 170% with the highest values observed within the deeper basins near the foot of the sill (stations C10 and C08).

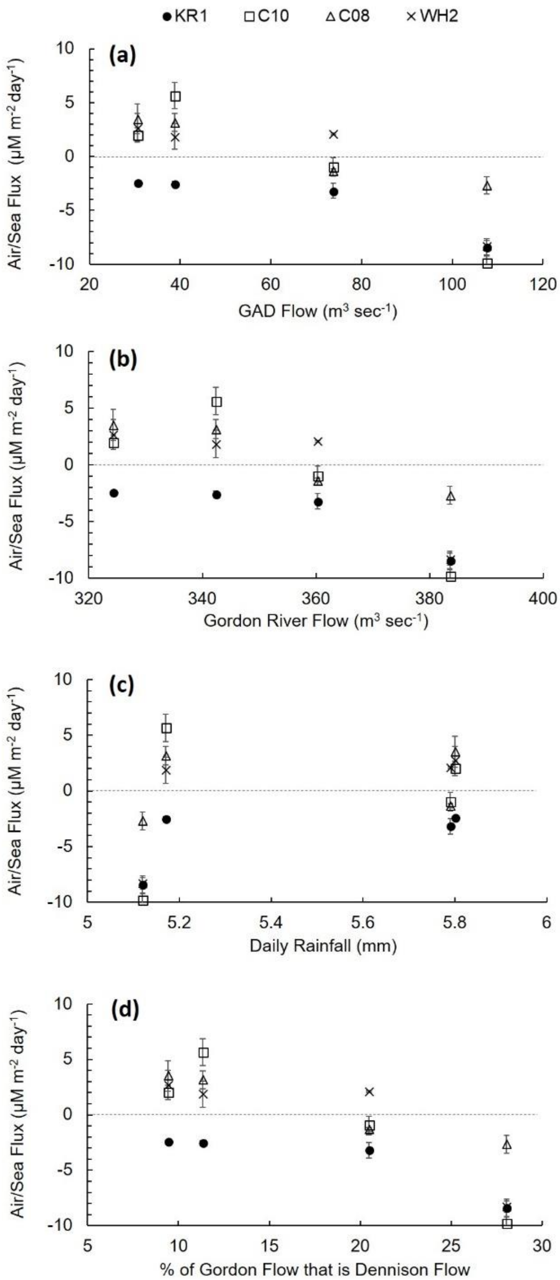

Figure 6: Mean Air/Sea Flux (µM m-2 day-1) versus A) Gordon above Dennison River flow (m3 day-1), B) estimated Gordon River flow (m3 day-1), C) daily rainfall (mm) (20 day mean), and D) % of estimated Gordon River flow this is accounted for by the Gordon above Dennison River gauge (proxy for hydroelectric dam release). Error bars indicate ± 1 standard error.

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In the harbour's subhalocline layer there is not enough light to support photosynthesis (**Hartstein *et al.*, 2019**; **Maxey *et al.*, 2017, 2020, and 2022**) and thus the main source of oxygen is advection from marine intrusions. N_2O producing microbes have been observed to populate this layer of the harbour (*see* **Da Silva *et al.*, 2021 and 2022**) and our observations of supersaturated N_2O in these layers show that those microbes are active. The linear relationship between AOU and ΔN_2O (slope = 0.0154; r = 0.596; p = 2.4 x $10^{-23;}$ **Figure 5C**) indicates that N_2O production occurs primarily through the ammonia oxidation (nitrification) pathway (**Yoshinari, 1976; Walter *et al.*, 2004; Brase *et al.*, 2017**). Our observations are on the lower end of the range reported nitrification slopes (see **Suntharalingam and Sarmiento, 2000; Brase *et al.*, 2017**) indicating a low yield of N_2O per mole O_2 consumed. These more modest yields are likely an artefact of mixing and loss dynamics such as basin water DO recharges from marine intrusions, and loss to aerobic respiration and loss to the atmosphere. This suggests that some portion of subhalocline pelagic oxygen demand in the harbour can be attributed to nitrifying microbes (albeit at a much lower rate compared to aerobic respiration). **Ji *et al.*, (2020)** also observed similar relationships in the Saanich Inlet, a seasonally anoxic fjord-like estuary in British Columbia, but in that system anoxic conditions are more persistent (**Bourbonnais *et al.*, 2013; Manning *et al.*, 2010**) compared to Macquarie Harbour (**Maxey *et al.*, 2022**). Deep-water renewal / marine intrusions have been hypothesized to stimulate N_2O production in the Saanich Inlet (**Capelle *et al.*, 2018; Michiles *et al.*, 2019**; **Ji *et al.*, 2020**), and Baltic Sea (**Walter *et al.*, 2006**) and may also be stimulating it in Macquarie Harbour as well. In the Baltic Sea, **Walter *et al*. (2006)** and **Myllykangas *et al.* (2017)** observed enhanced N_2O production in areas receiving significant marine intrusions. Positive correlations between AOU and ΔN_2O observed in western Baltic Sea waters (**Walter *et al.*, 2006**) along with mean (11-year; 2006-2017) seasonal variations in DO and N_2O observed through the water column at the Boknis Eck Time-Series Station (Eckernförde Bay, Southwest Baltic Sea) indicate a tight coupling between DO supply and N_2O production (presumably by nitrification) / consumption (presumably by denitrification) pathways in that area (**Ma *et al.*, 2019**). The reintroduction of marine water on the upstream side of a dam in the Nakong River, South Korea was found to affect bottom water trapping (stagnation), DO conditions, N process rates, process specific gene abundances, and subsequently the fate of N in that system (**Huang *et al.*, 2024**). Marine intrusions primarily refresh the DO supply adjacent to the sill in Macquarie Harbour (near station C10) and since we also observed a positive correlation between AOU and ΔN_2O they offer a possible explanation for the higher subhalocline N_2O concentrations observed there.

We conceptualize that during periods of high river flow, the surface water lens thickens and transports water undersaturated with N_2O quickly across the harbour surface and out of Hells Gates inlet. Some N_2O from the oversaturated subhalocline water is entrained in the surface lens (diapycnal flux) and transported out of the system in its dissolved form. During periods of low river flow, the surface lens is thinner and residence times longer (**Andrewartha and Wild-Allen 2017; Maxey *et al.*, 2022**). N_2O from the oversaturated subhalocline water then diffuses through the surface layer and escapes into the atmosphere in its gaseous form (**Figure 7**). Our estimates of diapycnal flux indicate that the mass transport from subhalocline waters is smaller (~2x smaller) than the air/sea flux supporting this idea. This conceptual model suggests that the harbour surface lens functions as a sink for both atmospheric N_2O and N_2O generated in the subhalocline layer during high flow periods (**Figure 7**).

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Previous work in Australian estuaries with pristine catchments (like Macquarie Harbour) has shown that many tend to function as a sink for atmospheric N_2O (**Maher *et al.*, 2016; Wells *et al.*, 2018**). Our study adds the caveat that source sink dynamics may also depend on factors controlling river flow in deeper stratified systems. Despite the advancements made to date, many of the deeper estuaries in Chile, Australia and New Zealand are lacking descriptions of N_2O source sink dynamics (*e.g.* Bathurst Harbour, Tasmania; fjords of South Island New Zealand; estuaries on Stewart Island New Zealand). Given that these systems have relatively pristine catchments they offer an opportunity to better understand natural fjord-like estuarine responses to the climate drivers of N_2O dynamics. Mesoscale climate oscillations (*i.e.* the Southern Annular Mode; SAM; North Atlantic Oscillation; NAO) have been shown to affect rainfall, river flow, and dissolved oxygen concentrations in this and other fjord-like estuaries (**Maxey *et al.*, 2022**; **Austin and Inall, 2002)**. In Western Tasmania, SAM in its positive phase results in increased orographic rainfall and a greater propensity for higher river flow, possibly tilting the source and sink balance to net N_2O uptake during these periods.

Climate change predictions for Tasmania's West Coast (which includes the Macquarie Harbour catchment) indicate that the region will experience a more extreme precipitation regime with increased winter precipitation and decreased summer precipitation (**Grose *et al.*, 2010; Bennett *et al.*, 2010**). If these future predictions result in more extreme seasonality in Gordon River flow, then the harbour may respond in kind with a larger variation in N_2O source and sink dynamics *i.e.* larger N_2O sink in winter and N_2O source in summer. However, given that the river flow is somewhat regulated by the hydroelectric dam, our study suggests that flow regulation has the potential to augment harbour N_2O emissions. Releasing water during extreme low rainfall periods might allow N_2O slowly accumulating in subhalocline waters to be released in the exported surface lens.

It is well established that fjord and fjord-like estuaries are important sites of C burial (**Smith *et al.*, 2015; Bianchi *et al.*, 2018, 2020**). This study supports the idea that they can also be important sites of N_2O sequestration. Macquarie Harbour air/sea flux estimates are similar in magnitude to observations made in other stratified estuaries and enclosed seas such as the Reloncaví Estuary, Chile (**Yevenes *et al.*, 2017**) and Eckernförde Bay, Germany (**Ma *et al.*, 2019**) (**Table A1**). Macquarie Harbour, however, was observed to have lower fluxes of N_2O into the atmosphere than other river dominated, but not fjord-like, estuaries (Elbe River, Germany; **Schulz *et al.*, 2023**) including those on the Australian mainland's east coast (**Wells *et al.*, 2018**).

Fjord and fjord-like estuaries are defined by their strong stratification and sensitivity to freshwater inputs. With climate change, rainfall patterns are expected to become more extreme and thus alter the river flow, and subsequently N_2O source sink dynamics in these systems on a global scale. In systems that are expected to experience increasingly drier conditions they may shift from net sinks of N_2O to sources, and further perpetuate the accumulation of N_2O in the atmosphere.

5. Conclusions

In summary, river flow, and specifically river flow driven by hydroelectric dam release, significantly affects both surface water N_2O concentrations and air/sea flux in Macquarie Harbour. Importantly, when river flow is low

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most of the harbour emits N_2O to the atmosphere. When river flow is high most of the harbour removes N_2O from the atmosphere and intercepts the diapycnal flux exporting N_2O to the ocean in its dissolved form.

N_2O is continually supersaturated below the halocline and the relationship between AOU and ΔN_2O indicate that the main N_2O generation process is nitrification. Climate change is predicted to result in wetter winter / drier summers for the Tasmanian West Coast, which may result in augmented N_2O air/sea fluxes.

These represent the first descriptions of N_2O spatiotemporal distribution, estimated air/sea flux, estimated diapycnal flux, and N_2O production pathways in this system.

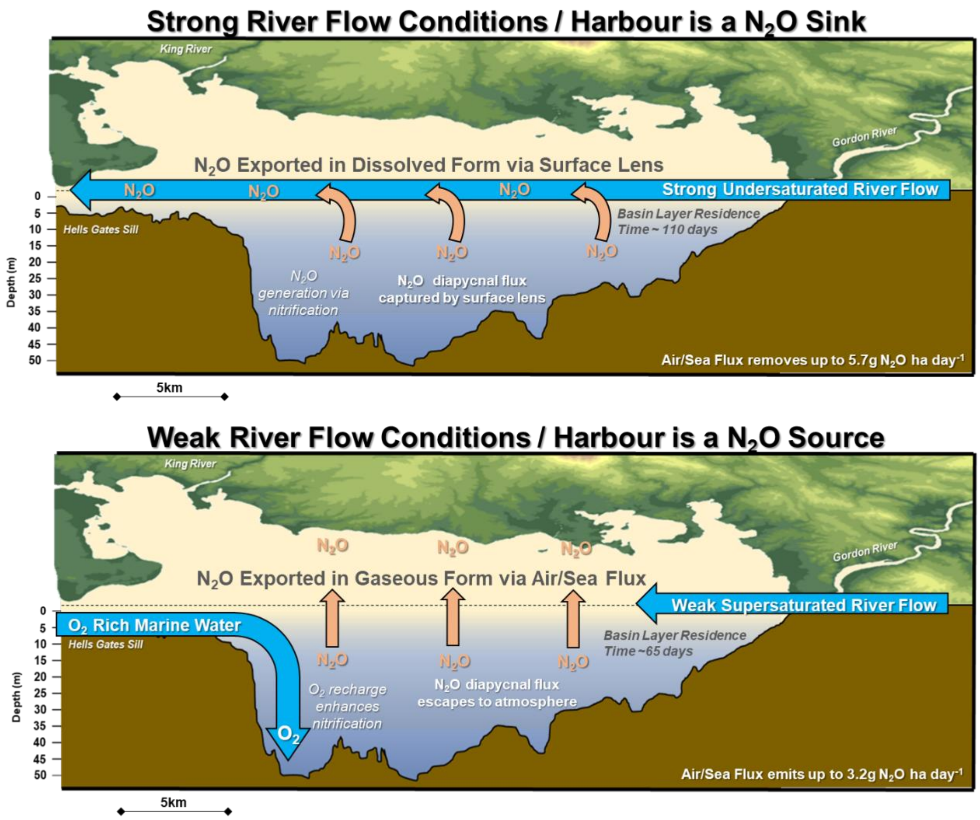

Figure 7: Conceptual model of Macquarie Harbour's N_2O dynamics. The top diagram depicts the capture of N_2O generated in the subhalocline during strong river flow conditions. Here N_2O is exported from the harbour in its dissolved form via undersaturated surface flows from the harbour to the ocean. The bottom diagram depicts the efflux of N_2O from the harbour surface during low flow conditions. Note that during these conditions the surface flows are weak and generally supersaturated with N_2O permitting its escape in gaseous form to the atmosphere.

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6. Appendix

Table A1: N_2O fluxes and observed ranges of mean (± standard deviation) N_2O concentration / saturation from both fjord-like / river dominated estuaries around the globe and estuaries in Australia.

Location	System Type	Measurement Depth Range	Mean Sea-to-Air N_2O flux uMol N_2O m^{-2} day^{-1}	Min and Max Sea-to-Air N_2O flux uMol N_2O m^{-2} day^{-1}	Mean N_2O Concentration (and Saturation) nM N_2O (and %)	Min and Max N_2O Concentration (and Saturation) nM N_2O (and %)	Reference
Macquarie Harbour, Western Tasmania, Australia	Fjord-like Estuary	2m to 45m	-09.83 ± 0.67 to 05.65 ± 1.22	-10.82 to 7.73	11.7 ± 1.6 (121.8 ± 17.8)	7.87 to 17.12 (81 to 174)	*This Study*
Reloncaví Estuary, Chile	Fjord-like Estuary	0m to 5m	0.86 ± 2.28	-1.58 to 5.60	11.8 ± 1.70 (111 ± 18.3)	8.34 to 14.5 (80 to 140)	Yevenes *et al.*, 2017
Reloncaví Estuary, Chile	Fjord-like Estuary	10m to 200m	-	-	14.5 ± 1.73 (145 ± 17.7)	10.5 to 17.0 (11 to 170)	Yevenes *et al.*, 2017
Chiloé Interior Sea, Chile	Fjord-like Estuary	0m to 200m	1.08 ± 1.41	-0.18 to 3.19	12.6 ± 2.36 (121 ± 17.5)	8.81 to 21.1 (87 to 160)	Yevenes *et al.*, 2017
Europa Sound, Magellanic Region, Chile	Fjord-like Estuary	1m to 10m	-15.22 to -0.81	-	11.9 ± 5.7 to 12.7 ± 1.0	-	Farías *et al.*, 2018
Concepción Channel, Magellanic Region, Chile	Fjord-like Estuary	1m to 150m	0.69 to 7.70	-	13.6 ± 1.1 to 17.0 ± 0.02	-	Farías *et al.*, 2018
Sarmiento Channel, Magellanic Region, Chile	Fjord-like Estuary	1m to 10m	2.07 to 12.53	-	13.1 ± 0.1 to 16.5 ± 0.3	-	Farías *et al.*, 2018
Estero Peel, Magellanic Region, Chile	Fjord-like Estuary	1m to 10m	0.11 to 2.01	-	13.1 ± 0.2 to 13.5 ± 0.5	-	Farías *et al.*, 2018
Estero Calvo, Magellanic Region, Chile	Fjord-like Estuary	1m to 10m	0.04	-	13.9 ± 0.8	-	Farías *et al.*, 2018
Estero Amalia, Magellanic Region, Chile	Fjord-like Estuary	1m to 100m	-0.08	-	14.2 ± 1.7	-	Farías *et al.*, 2018
Estero las Montañas, Magellanic Region, Chile	Fjord-like Estuary	1m to 10m	-2.95	-	9.69 ± 1.6	-	Farías *et al.*, 2018
Smyth Channel, Magellanic Region, Chile	Fjord-like Estuary	1m to 300m	1.07 to 11.2	-	14.3 ± 0.4 to 16.0 ± 0.5	-	Farías *et al.*, 2018
Última Esperanza Sound, Magellanic Region, Chile	Fjord-like Estuary	1m to 10m	-3.7 to 10.4	-	12.1 ± 1.1 to 13.7 ± 0.07	-	Farías *et al.*, 2018
Almirante Montt Gulf, Magellanic Region, Chile	Fjord-like Estuary	1m to 150m	15.6	-	21.0 ± 5.7	-	Farías *et al.*, 2018
Kirke Channel, Magellanic Region, Chile	Fjord-like Estuary	1m to 10m	0.12 to 8.19	-	13.3 ± 0.1 to 15.4 ± 0.4	-	Farías *et al.*, 2018
Union Channel, Magellanic Region, Chile	Fjord-like Estuary	1m to 10m	22.1	-	16.7 ± 0.8	-	Farías *et al.*, 2018
Union Sound, Magellanic Region, Chile	Fjord-like Estuary	1m to 10m	2.86	-	14.8 ± 0.8	-	Farías *et al.*, 2018
Western Magellan Strait, Magellanic Region, Chile	Fjord-like Estuary	1m to 10m	143	-	15.71	-	Farías *et al.*, 2018
Eastern Magellan Strait, Magellanic Region, Chile	Fjord-like Estuary	1m	36.3	-	16.4	-	Farías *et al.*, 2018
San Gregorio Cape, Magellanic Region, Chile	Fjord-like Estuary	1m	24.8	-	12.07	-	Farías *et al.*, 2018
Otway Center Sound, Magellanic Region, Chile	Fjord-like Estuary	1m	35.5	-	11.4	-	Farías *et al.*, 2018
Magdalena North Channel, Magellanic Region, Chile	Fjord-like Estuary	1m	-0.22	-	11.4	-	Farías *et al.*, 2018
Chasco Sound, Magellanic Region, Chile	Fjord-like Estuary	1m	6.81	-	16.01	-	Farías *et al.*, 2018
Cockburn West Channel, Magellanic Region, Chile	Fjord-like Estuary	1m	6.18	-	14.47	-	Farías *et al.*, 2018
Saanich Inlet, British Columbia, Canada	Fjord-like Estuary	10m to 200m	2.3 ± 2.5 to 3.9 ± 2.9	-	14.7	<0.5 to 37.4	Capelle *et al.*, 2018
Saanich Inlet, British Columbia, Canada	Fjord-like Estuary	Surface to 110m	11.3 to 20.4	-	-	-	Cohen 1978
Elbe River Estuary, Germany	Well-Mixed River Dominated Estuary	1.2m	-	26.0 ± 23.5 to 100.7 ± 101.2	-	(161 ± 53.6) to (243 ± 141.6)	Schulz *et al.* 2023
Eckernförde Bay, Boknis Eck Time Series Station, Baltic Sea, Germany	Enclosed Sea	1m to 25m	3.5 ± 12.4	-19.0 to 105.7	(111± 30)	(56 to 314)	Ma *et al.*, 2019
Eckernförde Bay, Boknis Eck Time Series Station, Baltic Sea, Germany	Enclosed Sea	1m to 25m	-	-	10 to 17	-	Walter *et al.*, 2006
Baltic Sea, Germany	Enclosed Sea	110m	5 -11	-	14 to 1523	-	Rönner 1983

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Gotland Basin, Baltic Sea, Germany	Enclosed Sea	90m	-	-	13	0 to 126 (0 to 450)	Brettar and Rheinheimer 1991
Northwest Shelf, Black Sea	Enclosed Sea	-	1.6 to 4.4	-	6.5 to 8	-	Amouroux *et al.*, 2002
Deep Basin, Black Sea	Enclosed Sea	70m	3.1 to 5.2	-	7.5 to 10.2	-	Amouroux *et al.*, 2002
Cariaco Basin, Venezuela	Coastal Basin	Surface to 400m	-	-	4.4 to 5.5	-	Hashimoto *et al.*, 1983
Guadalquivir Estuary, Gulf of Cadiz, Spain	River Dominated Estuary	2m	18.7 ± 33.6	-	20.6 ± 24.3	-	Sierra *et al.*, 2020
Guadalquivir Estuary, Gulf of Cadiz, Spain	River Dominated Estuary	2m	0.3 ± 0.5	-	6.7 ± 0.4	-	Sierra *et al.*, 2020
Guadalquivir Estuary, Gulf of Cadiz, Spain	River Dominated Estuary	2m	0.9 ± 21.6	-	7.3 ± 15.4	-	Sierra *et al.*, 2020
Noosa River Estuary, Eastern Australia	River Dominated Estuary	0.5m to 9.6m	-14.24 ± 14.02	-57.72 to 22.20	6.99 ± 0.43 (97 ± 2.2)	5.92 to 7.95 (90 to 103)	Wells *et al.*, 2018
Mooloolah River Estuary, Eastern Australia	River Dominated Estuary	0.5m to 6.8m	-7.33 ± 7.25	-48.76 to 16.31	6.74 ± 0.64 (97 ± 3.8)	5.19 to 7.71 (82 to 112)	Wells *et al.*, 2018
Maroochy River Estuary, Eastern Australia	River Dominated Estuary	0.5m to 8.2m	51.33 ± 55.3	-34.94 to 179.64	8.4 ± 1.50 (113 ± 16.7)	6.07 to 12.93 (92 to 163)	Wells *et al.*, 2018
Pine River Estuary, Eastern Australia	River Dominated Estuary	0.5m to 10.1m	17.10 ± 39.44	-33.22 to 145.50	7.1 ± 0.76 (102 ± 6.24)	6.05 to 8.57 (93 to 117)	Wells *et al.*, 2018
Brisbane River Estuary, Eastern Australia	River Dominated Estuary	0.5m to 23.9m	209.54 ± 143.59	15.42 to 662.62	9.8 ± 1.36 (133 ± 9.9)	6.75 to 12.75 (105 to 158)	Wells *et al.*, 2018
Middle Reach, Brisbane River Estuary, Eastern Australia	River Dominated Estuary	Surface	14.5 ± 1.19	5.4 ± 0.34 to 25.2 ± 1.87	-	13.1 to 17.9 (160 to 250)	Sturm *et al.*, 2017
Lower Reach, Brisbane River Estuary, Eastern Australia	River Dominated Estuary	Surface	6. ± 0.51	3.7 ± 0.85 to 9.1 ± 1.19	-	9.2 to 12.7 (125 to 410)	Sturm *et al.*, 2017
Oxley Creek, Eastern Australia	River Dominated Estuary	2.1m to 13.1m	210.59 ± 60.23	91.54 to 280.16	11.7 ± 1.34 (156 ± 19.7)	9.65 to 14.89 (139 to 199.7)	Wells *et al.*, 2017
Nerang River Estuary, Eastern Australia	River Dominated Estuary	0.5m to 6.8m	-0.62 ± 20.87	-67.98 to 45.92	6.73 ± 0.43 (100 ± 4.3)	5.99 to 7.79 (88 to 109)	Wells *et al.*, 2018
Logan River Estuary, Eastern Australia	-	0.5m to 14.4m	110.00 ± 153.55	-54.48 to 796.00	9.3 ± 2.36 (127 ± 27.5)	5.54 to 14.8 (81 to 191)	Wells *et al.*, 2018
Albert River Estuary, Eastern Australia	-	1.1m to 15.7m	90.05 ± 73.32	-9.50 to 264.25	10.10 ± 2.24 (131 ± 29.8)	7.32 to 15.1 (98 to 205)	Wells *et al.*, 2018
Darwin Creek, Australia	Mangrove Creek	~1m	-0.12	-	6.3 (98.9)	6.0 to 6.8 (95 to 104)	Maher *et al.*, 2016
Hinchinbrook Creek, Australia	Mangrove Creek	~1m	-3.43	-	6.1 (83.3)	5.6 to 6.8 (75 to 91)	Maher *et al.*, 2016
Melbourne Creek, Australia	Mangrove Creek	~1m	-1.33	-	7.9 (96.6)	6.9 to 9.1 (86 to 115)	Maher *et al.*, 2016
Morton Bay Creek, Australia	Mangrove Creek	~1m	-3.19	-	5.1 (77.4)	3.4 to 6.6 (50 to 105)	Maher *et al.*, 2016
Seventeen Seventy Creek, Australia	Mangrove Creek	~1m	-1.75	-	7.7 (94.3)	7.1 to 8.9 (88 to 106)	Maher *et al.*, 2016
Brisbane River, Australia	-	-	-	-	(285)	(135 to 435)	Musenze *et al.*, 2014
Coffs Creek, Australia	-	-	-	-	(219 ± 37)	(53 to 386)	Reading *et al.*, 2017
Coffs Creek, Australia	-	-	-	-	(266.5 ± 128)	(86 to 678)	Reading *et al.*, 2020
Boambee Creek, Australia	-	-	-	-	(197.1 ± 75)	(87 to 329)	Reading *et al.*, 2020
Bonville Creek, Australia	-	-	-	-	(183.7 ± 65)	(78 to 310)	Reading *et al.*, 2020
Pine Creek, Australia	-	-	-	-	(194.1 ± 65)	(79 to 382)	Reading *et al.*, 2020
Yarra River, Australia	Salt Wedge Estuary	-	-	-	(135.9 ± 31)	-	Tait *et al.*, 2017

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8. Data Availability

This data set is available upon request

9. Author Contributions

Johnathan Daniel Maxey – *Conceptualization, Field Collection, Analytical Methodology, Data Analysis, Writing – Original Draft, Writing – Review & Editing*

Neil David Hartstein – *Conceptualization, Field Collection, Analytical Guidance, Writing – Review & Editing, Funding*

Hermann W. Bange – *Conceptualization, Analytical Methodology, Data Analysis, Writing – Review & Editing*

Moritz Müller – *Conceptualization, Field Collection, Analytical Guidance, Writing – Review & Editing*

10. Competing Interests

HWB serves on the editorial board for Biogeosciences. The authors declare that they have no other conflicts of interest.

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