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Ineffective artificial mouth-breaching practices and altered hydrology confound eutrophic symptoms in a temporarily closed estuary

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ABSTRACT

Context. Artificial breaching of intermittently closed estuaries has become more frequent in the face of global-change pressures. **Aims.** This study aimed to determine whether the ecological health of the Great Brak Estuary has been affected by the prolonged loss of marine connectivity arising from below-average inflow and failed breaching attempts. **Methods.** We characterised primary eutrophic symptoms (inorganic nutrients, dissolved oxygen, microalgae) typical of the various mouth states, i.e. open, closed and semi-closed. **Key results.** Initially, low inflow and closed mouth conditions facilitated the widespread occurrence of macroalgal blooms (*Cladophora glomerata*). Phytoplankton bloom conditions ($>20\ \mu g$ Chl-*a* L^{-1}) ensued only in response to favourable hydrodynamic conditions (e.g. increased water residency, halocline formation) and increased nutrient availability from fluvial sources and macroalgal dieback. These blooms occurred in brackish conditions and comprised numerous taxa, including *Cyclotella atomus* var. *marina*, *Cryptomonas* sp. and *Prorocentrum cordatum*. Widespread hypoxia (<2 mg L^{-1}) occurred during the semi-closed mouth phase because of the reduced flushing potential associated with the preceding high flow conditions. **Conclusions.** Global-change pressures and ineffective breaching practices will promote eutrophic conditions in intermittently closed estuaries in the future. **Implications.** Allocating sufficient environmental flows is key to preventing ecosystem degradation.

Keywords: eutrophication, harmful algal blooms, hypoxia, macroalgae, microphytobenthos, mouth management, phytoplankton, water quality.

Introduction

Temporarily closed estuaries (TCEs) are highly threatened by the demand for freshwater resources (socio-economic versus environmental), pollution and climate change (Stein *et al.* 2021; van Niekerk *et al.* 2022*a*, 2022*b*). They require seasonal changes in mouth status (open or closed) to maintain healthy ecological processes, which are mainly driven by fluvial freshwater inputs (Van Niekerk *et al.* 2019*a*; Adams and Van Niekerk 2020). Three dominant hydrodynamic states can be defined for TCEs, depending on the rate and volume of freshwater inflow and tidal exchange, i.e. open, semi-closed (overtopping or overwashing) and closed (Snow and Taljaard 2007; Van Niekerk *et al.* 2020*a*). Modifications in rivers and estuaries, including flow obstructions, change the natural baseflow to estuaries, causing premature mouth closure (Van Niekerk *et al.* 2013). This is further exacerbated when the supply of fluvial or marine sediments is prominent, creating a sand bar at the mouth inlet (Whitfield and Bate 2007; McSweeney *et al.* 2017). The closed phase is essential to the ecological functioning of estuaries, but the open phase promotes the formation of intertidal habitat, enhances nursery functionality and improves water quality (Adams and Van Niekerk 2020; Chilton *et al.* 2021). Longer periods of closure can cause habitat loss, decreased species richness and eutrophic conditions, all of which act synergistically to

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hinder the provision of important ecosystem services (e.g. Terörde and Turpie 2013; Lemley *et al*. 2014*a*, 2018, 2021; Nunes and Adams 2014; Young *et al*. 2022).

Microalgae form the base of the food-web in most estuaries, contributing up to 50% of the total estuarine autochthonous primary production (Underwood and Kromkamp 1999; Whitfield and Bate 2007). Temporal and spatial variations in microalgal composition, biomass and production can be triggered by shifts in mouth states (Perissinotto *et al*. 2002; Anandraj *et al*. 2007). Freshwater inflow directly influences the residence time of nutrients and the accumulation of microalgal biomass, through increasing or decreasing flushing rates and maintaining or creating openmouth conditions that facilitate marine connectivity (McSweeney *et al*. 2017; Brooker and Scharler 2020; Adams and Van Niekerk 2020; Van Niekerk *et al*. 2022*a*). Mouth closure occurs when input of wave-deposited marine sediment exceeds freshwater input, and often leads to increases in microalgal biomass (Dalu *et al*. 2018; Lemley and Adams 2020). In the presence of anthropogenic stressors, this state promotes the production of opportunistic algal species, predominantly in the form of fast-growing macroalgal and phytoplankton species. Dense accumulations of phytoplankton can cause harmful algal blooms (HABs) that can have detrimental consequences on estuarine ecosystem functionality. This includes direct toxicity (bioaccumulation), oxygen depletion of bottom waters (associated with decay), suffocation of faunal communities, shifts in food-web structuring and a loss of important nursery habitats (e.g. submerged macrophytes) (e.g. Lemley and Adams 2019).

In light of global climate change and sea-level rise, semi-arid regions such as South Africa, Australia, USA (e.g. California, Texas) and coastal Mediterranean countries (e.g. Spain, Portugal), for example, are expected to become drier and receive less rainfall (Van Niekerk *et al* 2022*b*). As a result, it is important to manage estuaries with future climate variability in mind to maintain ecosystem functionality and ensure continued provision of socio-economic benefits (Brooker and Scharler 2020). Numerous studies have highlighted the potential for enhanced estuarine degradation (e.g. HABs, mouth closure, hypoxia) in response to various climate-change scenarios (e.g. drought periods, warming) (O'Neil *et al*. 2012; Wetz and Yoskowitz 2013; Lemley and Adams 2020). However, the demand for freshwater continues to increase because of human population growth, urbanisation and industrialisation (Schlacher and Wooldridge 1996; Cloern *et al*. 2016). The mouth inlets of estuaries subjected to urban development are often managed to mimic the natural conditions of the estuary, with the key objective being to reduce the flood risk to low-lying developments. Artificial mouth breaching (i.e. mechanical removal of sand bar) has been applied in many estuaries globally as a means of maintaining water quality (Lill *et al*. 2012; Netto *et al*. 2012; Human *et al*. 2016; Suari *et al*. 2019; Mayjor *et al*. 2023). As such, artificial breaching practices in estuaries are likely to become more frequent to ensure functional estuarine systems and economic prosperity.

The temporarily closed Great Brak Estuary situated along the southern coast of South Africa has experienced more frequent periods of prolonged mouth closure following the construction of an upstream dam impoundment in 1988 (Slinger *et al*. 2005). As a result of flow modification and vulnerability of low-lying developments in the floodplain, artificial mouth breaching of the estuary is common practice (Department of Water Affairs 2009; Van Niekerk *et al*. 2018). In South Africa, detailed mouth management plans for TCEs are implemented to maintain or improve the health of modified estuaries towards a reference condition, and to protect the surrounding properties and infrastructure (Department of Water Affairs 2009; Van Niekerk *et al*. 2018; NCC Environmental Services and Anchor Environmental Consultants, unpubl. data). Consequently, a mouth management plan was designed to maintain annual open-mouth states (i.e. September to April) geared towards imitating natural ecosystem functionality. However, in 2010, no water releases were made from the dam because of the persistent drought, which led to the mouth of the Great Brak Estuary remaining closed for almost 2 years. The prolonged mouth closure, combined with increased nutrient availability (*in situ* remineralisation), facilitates the proliferation of, and shifts between, macroalgae and phytoplankton within the estuary (Nunes and Adams 2014; Human *et al*. 2015; Lemley *et al*. 2015). Since 2016, the Great Brak catchment has again been experiencing below-average rainfall that has subsequently increased the frequency of prolonged periods of mouth closure and artificial breaching at lower water levels in the past few years (NCC Environmental Services and Anchor Environmental Consultants, unpubl. data). Failing to maintain annual open-mouth conditions can lead to a catastrophic decline in environmental health. The aim of this study was to determine whether the water quality and microalgal dynamics (i.e. primary eutrophication symptoms) have been affected by the loss of marine connectivity in the Great Brak Estuary, resulting from prolonged drought periods and failed breaching attempts. It is hypothesised that artificial breaching practices that facilitate prolonged open mouth conditions will improve estuarine water quality and prevent eutrophic conditions (e.g. hypoxia, phytoplankton blooms). Globally, climate change and overpopulation have put pressure on our natural resources. Studies that can link detrimental conditions (e.g. drought, eutrophication) to these stressors are key in conserving healthy ecosystems and mitigating the effects of a changing climate.

Materials and methods

Study site

The Great Brak Estuary (34°03′23″S, 22°14′18″E) is classified as a large temporarily closed estuary (105 ha), located along the warm-temperate southern coast of South Africa (Van Niekerk *et al.* 2019*b*, 2020*a*). It drains a small, forested catchment area of 192 km^2 (Department of Water Affairs 2009). The estuary is 6 km long and is significantly developed on either side of the banks (Fig. 1). The area around the mouth is surrounded by a rocky headland on the east and a sand berm on the west. Sediment naturally accumulates at the sand berm and around the mouth. Upstream from the mouth, the estuary opens up into a lagoon, which contains a developed island. The sand berm generally prevents water from flowing out of the estuary on the east of the island (Van Niekerk *et al.* 2018). There are five bridges and a causeway in the Great Brak Estuary, two of which have a significant impact on estuarine flow. They include the Island Bridge in the lower reaches, and the Charles Searle Bridge (Fig. 1; between Stations M4 and S6) in the upper reaches of the estuary. Additionally, a causeway in the upper reaches also obstructs tidal exchange and freshwater inflow during low-flow periods. Development also includes residential (formal and informal), agricultural and commercial zones.

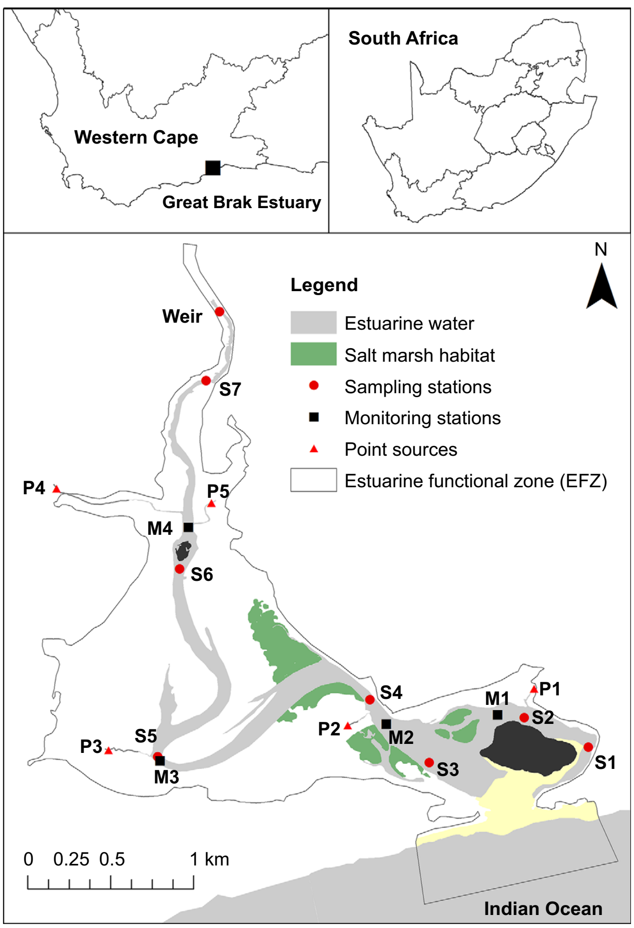

Fig. 1. Locality of the various sampling stations throughout the Great Brak Estuary, i.e. comprehensive survey sites (S1–S7 and Weir), intensive sampling sites (M1–M4) and point-source sites (P1–P5).

Run-off from a golf course enters the estuary in the lower reaches (Fig. 1; Station P1). The town attracts various recreational activities such as yachting, wind surfing, fishing, bird watching and swimming (NCC Environmental Services and Anchor Environmental Consultants, unpubl. data).

A mouth management plan was developed in response to the construction of the Wolwedans Dam (capacity 23×10^6 m^3) that was completed in 1990, although informal breaching had been implemented since the early 1800s (Franklin 1975). Artificial breaching occurs once a year at the start of spring (September–October) but can be delayed if dam levels are low and environmental flows are not available to maintain ecological objectives or socio-economic activities (Van Niekerk *et al.* 2018, 2022*a*). The aim is to maintain the open mouth conditions throughout spring and summer, and flow releases and re-breaching may be implemented from September to April, in cases of premature closure. Furthermore, emergency breaching can be implemented in response to floods or if the dam overflows. The dam is located 3 km upstream of the estuary tidal head (Fig. 1; Slinger *et al.* 2005). Studies have indicated that natural mouth breaching is no longer possible in the Great Brak Estuary, because of low-lying developments in the estuarine functional zone and freshwater abstraction from the Wolwedans Dam (Department of Water Affairs 2009; Van Niekerk *et al.* 2018). The estuary is currently rated as highly modified (Estuarine Health Index = D; Van Niekerk *et al.* 2019*b*). The system is rated as being of medium importance in terms of biodiversity, especially as it acts as an important plant habitat (33 species) and nursery area for various fish species (Department of Water Affairs 2009; Lemley *et al.* 2021). It is oligotrophic in its natural state, but its most recent classification indicated that it resides in a mesotrophic condition (Lemley *et al.* 2015). The area experiences a bimodal rainfall pattern with peaks in spring and autumn (Lemley *et al.* 2014*b*), while also being subject to periodical droughts and floods. The abiotic conditions within the estuary are highly dependent on the state of the mouth, and artificial mouth breaching is implemented to ensure connectivity with the marine environment during the key fish recruitment periods in spring and summer (Van Niekerk *et al.* 2018). Incidentally, the timing of mouth breaching coincides with the typical growth period of the filamentous macroalgae species *Cladophora glomerata*, which has been documented to proliferate in the estuary (Lemley *et al.* 2014*a*; Nunes and Adams 2014; Human *et al.* 2016). As such, open mouth conditions facilitated by mechanical breaching serve as a means of reducing the impacts of these events.

Sampling

The mouth of the Great Brak Estuary was breached on 17 November 2021, after being closed for approximately a year and was supervised by the Department of Water and

Sanitation (DWS), Department of Environmental Affairs and Development Planning (DEA&DP), Breede-Gouritz Catchment Management Agency (BGCMA), CapeNature and the Mossel Bay Municipality, according to the specification set aside in the Environmental Maintenance Management Plan (NCC Environmental Services and Anchor Environmental Consultants, unpubl. data). Three comprehensive spatial surveys were conducted in the Great Brak Estuary to characterise the abiotic and microalgal conditions typical of the various mouth conditions, i.e. 31 July 2021 (closed mouth), 26 November 2021 (open mouth) and 18 February 2022 (semi-closed mouth, i.e. overflowing on high tides). For the comprehensive surveys, eight sampling locations were sampled along the length of the estuary (Fig. 1; S1–S7 and Weir). These eight sites were referred to in terms of 'distance from mouth', which corresponded numerically to 0.6 km (S1), 1.0 km (S2), 1.6 km (S3), 2.15 km (S4), 3.5 km (S5), 4.85 km (S6), 6.0 km (S7) and 6.7 km (Weir). This component of the study investigated water quality, phytoplankton, microphytobenthos (MPB) and sediment properties at each site. Furthermore, five sites (i.e. P1–P5; see Fig. 1) were sampled during the three comprehensive surveys to characterise the potential point sources entering the estuary. In addition to the comprehensive spatial surveys, an intensive sampling program was conducted twice monthly from 23 September 2021 to 26 April 2022 (n = 16), to investigate water quality and phytoplankton-bloom dynamics. On each occasion, samples were collected at four fixed stations (M1–M4; Fig. 1).

Water level and flow

Long-term monitoring data from permanent gauging stations (Stations K2H002 and K2T004) were obtained from the Department of Water and Sanitation (DWS), South Africa. Freshwater inflow (m^3 s^{-1}) and water-level (above mean sea level, m MSL) data were available for the period of 1 January 1988–30 April 2022 and 2 May 1988–30 April 2022 respectively, i.e. approximately 34 years. Freshwater inflow data were recorded once per day and water level data were recorded every 6 min. Data for the last 10 years of the monitoring period (i.e. 1 January 2012–30 April 2022) were extracted to emphasise the hydrological conditions of the Great Brak Estuary during the recent drought period. The freshwater inflow conditions were classified as low (<25th percentile), normal (25–75th percentile) or high (>75th percentile) by using a percentile-based approach.

Abiotic parameters

Physico-chemical parameters were measured at the aforementioned sites with either a YSI ProDSS (comprehensive surveys) or HANNA HI98194 (intensive sampling) multiparameter meter. The measured parameters included salinity, dissolved oxygen (mg L^{-1}), pH and temperature (°C). Comprehensive sites (S1–S7 and Weir) were sampled at 0.5-m depth intervals from the surface to the bottom (depth range 0.5–5 m) of the water column. Secchi depths (m) were also measured at these sites. Intensive sites and point sources were measured at the surface of the water column. Water samples were collected using a weighted pop-bottle. Subsequently, water samples were gravity filtered through glass-fibre filters (Whatman GF/C) and then re-filtered through sterile 25-mm cellulose acetate 0.45-μm pore-size syringe filters. The Whatman GF/C filters were kept for phytoplankton biomass analysis (see below) and the syringe-filtered filtrates were placed into acid-washed polyethylene bottles and subsequently frozen. Once in the laboratory, filtrates were thawed and analysed for orthophosphate (PO_4^{3-}), ammonium (NH_4^+), total oxidised nitrogen (NO_x, i.e. NO_3^- + NO_2^-) and dissolved silica (SiO_2) by using a SEAL AutoAnalyzer 3 HR (SEAL Analytical, Inc., Germany) (Murphy and Riley 1962; Grasshoff *et al.* 1999). Additionally, nutrients were categorised as dissolved inorganic nitrogen (DIN = NH_4^+ + NO_x) and phosphorus (DIP = PO_4^{3-}).

Microalgal biomass and community composition

Phytoplankton biomass, measured as chlorophyll-*a* (Chl-*a*) concentration, was determined by filtering triplicate samples of a known volume of water (ranging from 250 to 500 mL) through 1.2-mm pore-sized glass-fibre filters (Whatman GF/C). The filters were then placed in aluminium foil and frozen prior to analysis. Once in the laboratory, Chl-*a* was extracted by placing thawed filters into glass scintillation vials with 10 mL of 95% ethanol (Merck 4111) for 24 h in a cold (~2°C), dark room. Thereafter, suspended particles were removed by re-filtering the extracts. The absorbance of the cleared extracts, before and after acidification with one drop of 1 N of HCl, was then determined using a Thermo Scientific GENESYS 10S UV–Vis spectrophotometer at a wavelength of 665 nm. The Chl-*a* concentrations (μg L^{-1}) were calculated using the equation of Snow *et al.* (2000), derived from Nusch (1980). Microphytobenthos (MPB) biomass was also determined using Chl-*a* concentration as a proxy. Triplicate subtidal samples from the top centimetre of sediment were collected at each site by using a Perspex twin corer with an internal diameter of 20 mm and placed into acid-washed polypropylene containers prior to being frozen. In the laboratory, the samples were thawed at room temperature, and extracted using 15 mL of 95% ethanol for 6 h in a cold dark room (1–2°C), before being re-filtered through glass-fibre filters (Whatman GF/C) to clear any suspended particles. The Chl-*a* concentration was then determined with the same method as described above for phytoplankton samples and expressed as milligrams of Chl-*a* per square metre. In conjunction with this, sediment from each site was dried at 105°C for 24 h and weighed, and subsequently placed in an ashing oven at 550°C for 12 h to determine the sediment organic fraction (%).

For phytoplankton community composition and enumeration, water samples (100 mL) were preserved with 1 mL of

25% glutaraldehyde solution The preserved water samples were prepared by adding two drops of Rose Bengal (Sigma–Aldrich Chemicals R3877) to a known volume (i.e. 10–50 mL) and allowed to settle for 24 h in 26.5-mm diameter Utermöhl settling chamber (Coulon and Alexander 1972). Phytoplankton were identified and counted using an inverted Leica DMIL phase contrast microscope at a magnification of 630×, with a minimum of 200 cells or frames counted for each sample (cells per mL; Snow *et al.* 2000). The cells were classified according to classes, including Bacillariophyceae, Chlorophyceae, Chlorodendrophyceae, Cryptophyceae, Cyanophyceae, Dinophyceae, Euglenophyceae and *Mesodinium* cf. *rubrum* (i.e. mixotrophic ciliate, class Litostomatea). The following databases were used to obtain recent taxonomical classifications and salinity preferences: World Register of Marine Species (see http://www.marinespecies.org), AlgaeBase (M. D. Guiry and G. M. Guiry, see http://www.algaebase.org) and Diatoms of North America (Spaulding *et al.* 2021).

Data analysis

Data analysis and graphical representations were created using R (ver. 4.1.2, R Foundation for Statistical Computing Vienna, Vienna, Austria, see https://www.r-project.org/). However, contour graphs were created using Grapher (ver. 6.0, Golden Software, LLC, see www.goldensoftware.com). Data were analysed for normality using the Shapiro–Wilks test. Means were then compared using one-way ANOVA or the Kruskal–Wallis test for the parametric and non-parametric data respectively. Post hoc analysis was used to compare data between the specific mouth states by using the Tukey test or the Bonferroni Dunn test for parametric and non-parametric data respectively. Pearson's correlations were analysed for all biotic and abiotic parameters. All statistical significances were reported when $P < 0.05$. All analyses were tested at a 95% confidence level.

Results

Hydrological variability

Over the period from 1988 to 2022, typical freshwater inflow rates to the Great Brak Estuary ranged from 0.008 to 0.148 $m^3 s^{-1}$ (Table 1; interquartile range). However, the freshwater inflow rate decreased in the last 20 years of the studied period, and even more so in the last 2 years leading to this study. The most extreme difference can be seen when analysing the high-flow conditions (i.e. 99th percentile), where flood peaks decreased by 69% in the last 10 years of the studied period, and by 84% when only the last 2 years of the period were considered, compared with the 34 years from 1988 to 2022. Drought conditions in the last 2 years (10th percentile) of this period have seen no inflow of freshwater (0 $m^3 s^{-1}$) to the estuary, whereas the conditions for the 34 years from 1988 to 2022 indicate a flow of 0.03 $m^3 s^{-1}$ (Table 1). Freshwater inflow (floods represented by spills over the dam wall and dam releases) also steadily decreased in occurrence and intensity during the last 10 years of the studied period, with a particular decline from 2016 into 2021 (Fig. 2). In an attempt to sustain an open mouth after breaching, intermittent freshwater releases from the Wolwedans Dam are initiated, as was the case after the artificial breaching event in November 2021. This can be seen by a sudden increase in flow rate starting at the end of 2021 (Fig. 2). The water level (m MSL) of the estuary reflects periods of mouth closure as a lower and more stable water level, whereas open mouth conditions show a sudden decline in the water level. When closed, the water level in the system is elevated above sea level, and after breaching, it initially drains and then becomes tidal (Fig. 3). Freshwater releases from the Wolwedans Dam are visible as an atypical increase in water level shortly before an artificial breaching and over neap tides to keep the mouth open. Pulse releases are also practised to flush open the mouth shortly after it closed, while the berm is still low. The general stability of

Table 1. Quantification of flow rate ($m^3 s^{-1}$) conditions for the Great Brak catchment, where low flow is ≤25th percentile, normal flow is 25–75th percentile and high flow is 75–99th percentile.

Percentile	Flow rate ($m^3 s^{-1}$) for a set time (years)			
	1988–2022	2002–2012	2012–2022	2020–2022
1st	0.000	0.000	0.000	0.000
10th	0.003	0.003	0.001	0.000
25th	0.008	0.008	0.005	0.002
50th	0.020	0.014	0.008	0.004
75th	0.148	0.167	0.027	0.008
90th	0.613	0.506	0.479	0.114
99th	22.270	9.303	6.959	3.644

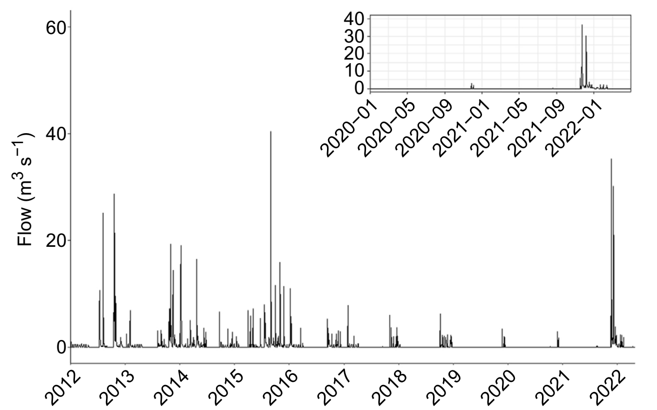

Fig. 2. Flow rate ($m^3 s^{-1}$) into the Great Brak Estuary starting in January 2012 to April 2022, with an inset showing the period of January 2020 to April 2022.

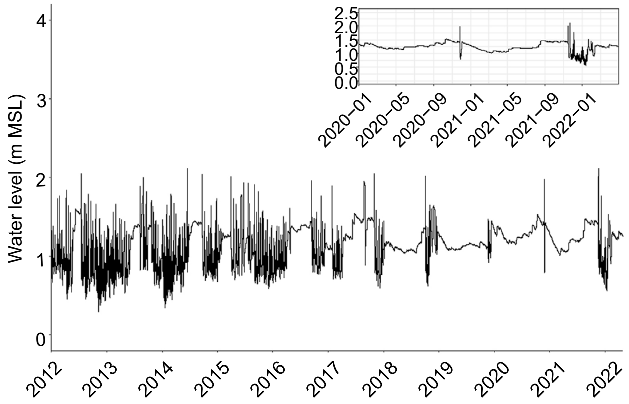

Fig. 3. Water level (m MSL) of the Great Brak Estuary starting in January 2012 to April 2022, with an inset showing the period of January 2020 to April 2022.

the water level in the last 2 years of the studied period indicates longer periods of mouth closure. This, together with the inefficacy of artificial breaching attempts, is highlighted by the brief periods of open mouth conditions (~4 days) observed subsequent to breaching in 2020 and 2021. However, during this study, despite initially closing 4 days after breaching on 21 November 2021, a natural breaching event occurred 2 days later at the onset of flood conditions (i.e. dam overflow).

Physico-chemistry and inorganic nutrients

Salinity gradients were different for each of the three mouth states during the comprehensive surveys (Fig. 4). During the closed mouth state (31 July 2021), mesohaline conditions occurred in the estuary (17.8 ± 0.9). Salinity was largely homogenous (range 13.9–18.5), with a slight horizontal salinity gradient decreasing towards the upper reaches. At the time of sampling, the mouth was closed for more than a year. Oligohaline conditions (2.3 ± 3.1) occurred after the artificial breaching (26 November 2021). The surface water was largely fresh, with more saline conditions (>6) confined to the deepest portion of the estuary (S5; depth 4.8 m). During the semi-closed mouth state (18 February 2022), euhaline conditions occurred with both horizontal and vertical salinity gradients present (19.6 ± 6.5). The upper reaches towards the weir were fresh (0.3), whereas the highest salinity occurred closest to the mouth (31.2) and at the deep Site 5 (31.8; 3.5 km from the mouth), both in the bottom waters. During the intensive sampling program, the estuary started off with mesohaline conditions during the closed phase (range 9.7–16.6), but once the mouth opened due to flood conditions, salinity decreased towards oligohaline conditions (Fig. 5). Large inter-site differences only occurred approximately 1 month after breaching, which corresponds to reduced freshwater inflow into the estuary (Fig. 2). Salinity then stabilised between sites when the mouth closed again (19.3 ± 0.8).

Water temperature displayed expected seasonal shifts, with the winter closed mouth state (13.5 ± 0.8°C) being considerably cooler than open (20.0 ± 1.3°C) and semi-closed (23.8 ± 1.6°C) conditions in spring and summer (Fig. 4). Similar observations were made during the intensive survey, with spring (19.4 ± 2.6°C) and autumn (20.2 ± 2.3°C) conditions being cooler than those recorded in summer (23.7 ± 2.5°C) (Fig. 5). In terms of spatial gradients, longitudinal temperature gradients were observed during the open (decreasing from mouth to upper reaches) and semi-closed (increasing towards upper reaches) states, whereas homogenous conditions were observed during closed mouth conditions.

Dissolved oxygen (DO) concentrations (Fig. 4) were categorised, by applying the 10th percentile (*sensu* Lemley *et al.* 2015), into 'well-oxygenated' (>5 mg L^{-1}), 'biologically stressful' (>2–≤5 mg L^{-1}), 'hypoxic' (>0–≤2 mg L^{-1}) and 'anoxic' (0 mg L^{-1}) conditions. During the closed mouth state, the water column was well-oxygenated (range 7.5–11.3 mg L^{-1}; 10th percentile 8.8 mg L^{-1}). Higher DO concentrations were recorded in the lower reaches and at the surface, than in the upper reaches and bottom waters. Dissolved oxygen concentrations recorded during open mouth conditions were lower (range 2.3–9.5 mg L^{-1}; 10th percentile 6.4 mg L^{-1}). The lowest value occurred at the deepest area (S5; 4.8 m; 3.5 km from mouth), whereas the highest value occurred at 6.0 km, and was homogenous with depth. Therefore, during the fluvially dominated open phase, horizontal stratification was less pronounced, and vertical stratification was more pronounced in deeper areas (>3 m). During the semi-closed mouth phase, DO concentrations were lowest (range 0.4–8.5 mg L^{-1}; 10th percentile 0.7 mg L^{-1}), reaching hypoxic conditions in the bottom waters of the middle and upper reaches. Vertical stratification was present, with higher DO occurring at the surface than in the bottom waters. During the intensive survey, DO concentrations generally showed the inverse to salinity (Fig. 5). The pattern indicates a significant drop in DO concentrations approximately a month after breaching (Fig. 2). The surface waters were typically well-oxygenated throughout, declining to a minimum of ~4 mg L^{-1}.

Inorganic nutrients (NH_4^+, NO_x, DIP, dissolved silica) were markedly different in magnitude and spatial distribution, among the three mouth phases (Fig. 6, 7). Dissolved inorganic nitrogen (DIN) and phosphorus (DIP) were categorised *sensu* Lemley *et al.* (2015), by applying the 80th percentile, into 'oligotrophic' (DIN 0–<0.1 mg L^{-1}; DIP 0–<0.01 mg L^{-1}), 'mesotrophic' (DIN ≥0.1–<1 mg L^{-1}, DIP ≥0.01–<0.1 mg L^{-1}) and 'eutrophic' (DIN >1 mg L^{-1}; DIP >0.1 mg L^{-1}). According to the above thresholds, DIN was classified as oligotrophic in the closed phase (0.03 mg L^{-1}), and mesotrophic in the open (0.1 mg L^{-1}) and semi-closed (0.2 mg L^{-1}) phases. The mean ammonium (NH_4^+) and NO_x concentrations were

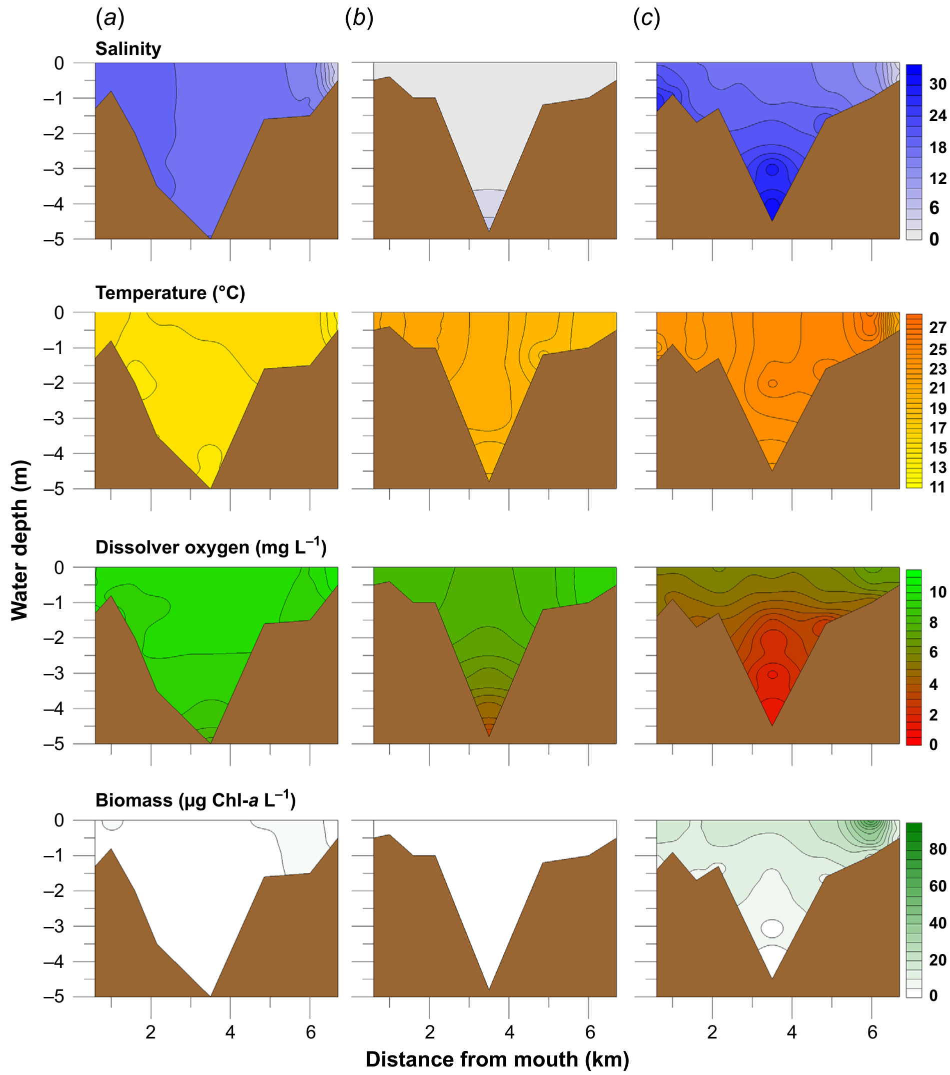

Fig. 4. Spatial distribution profiles of salinity, water temperature (°C), dissolved oxygen (mg L^{-1}) and phytoplankton biomass (Chl-*a*) observed during the comprehensive surveys, i.e. (*a*) closed (31 July 2021), (*b*) open (26 November 2021) and (*c*) semi-closed (18 February 2022) mouth states.

significantly different among the three mouth states (NH_4^+ $\chi^2 = 42.40$, $P < 0.001$, $n = 116$; and NO_x $\chi^2 = 150.5$, $P < 0.001$, $n = 116$). Ammonium concentrations generally increased from closed to open to semi-closed state; however higher mean concentrations were observed in the middle reaches during the closed phase. The NO_x concentrations showed a different pattern, where the highest mean concentrations were observed in the open phase. The higher concentrations were maintained in the lower reaches during the semi-closed phase, but the mean NO_x concentrations were much lower in the middle and lower reaches. The DIP concentrations showed mesotrophic conditions in all three mouth phases, i.e. closed (0.05 mg L^{-1}), open (0.03 mg L^{-1}) and semi-closed (0.04 mg L^{-1}) phases. The mean DIP in the closed phase differed significantly from those in the open and semi-closed phases ($\chi^2 = 44.88$, $P < 0.001$, $n = 116$), indicating significant changes in DIP as a result of mouth breaching. Last, dissolved silica concentrations were lowest during the closed mouth state compared with periods of marine connectivity ($\chi^2 = 73.88$, $P < 0.001$, $n = 116$). Additionally, dissolved silica was generally higher in the upper reaches than in the lower reaches, with the exception of the open mouth state where the reverse pattern occurred.

Monitoring of point-source sites (Supplementary Table S1) highlighted that only the station in the upper reaches (P4; see Fig. 1) provides a constant source of inflow to the estuary. Dissolved inorganic nutrients measured at Station P4 exhibited high DIP concentrations (>0.1 mg L^{-1}) throughout the study period. Furthermore, DIN concentrations were low (<0.03 mg L^{-1}) during the closed and semi-closed mouth states (i.e. low-flow periods) and increased to moderate

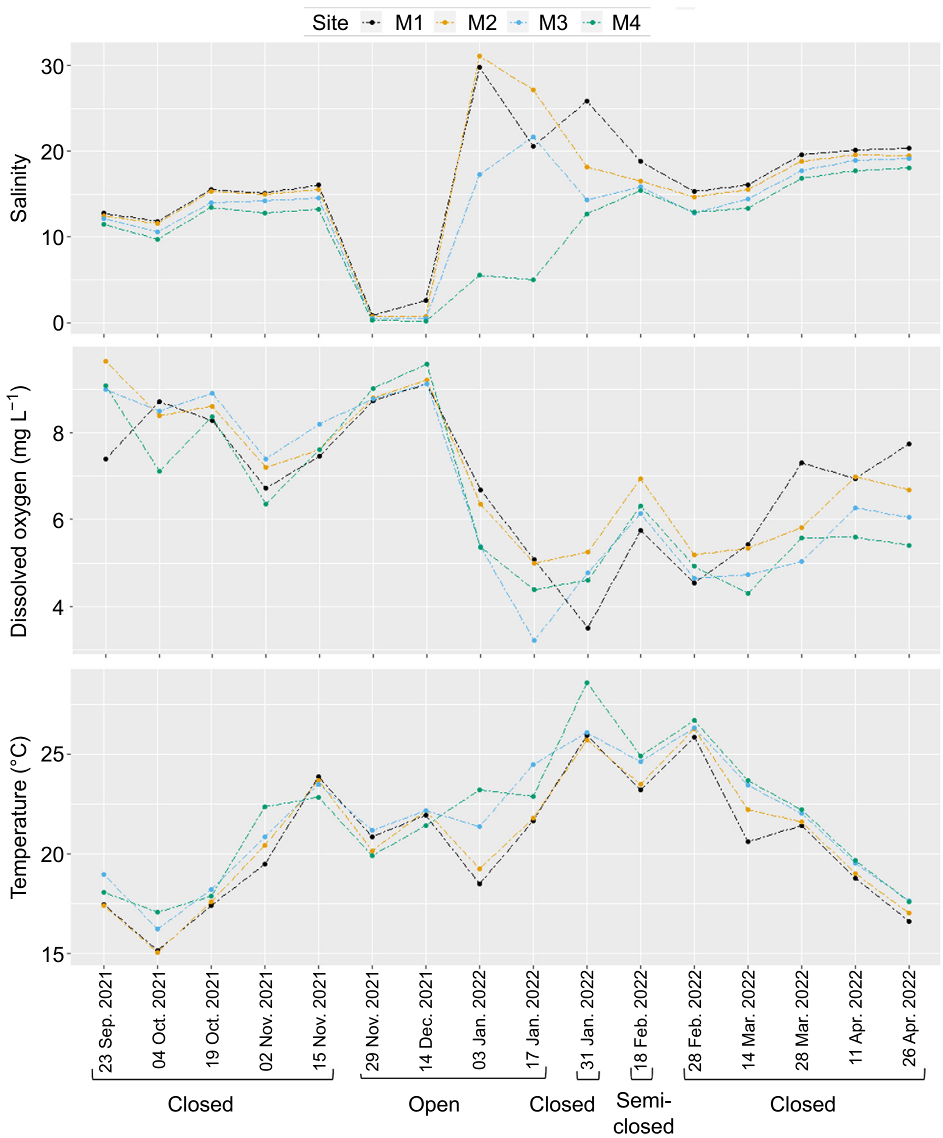

Fig. 5. Salinity, dissolved oxygen (mg L^{-1}) and water temperature (°C) during the intensive sampling program (M1–M4). The mouth state corresponding to each time period is displayed on the *x*-axis.

concentrations (0.22 mg L^{-1}) at the onset of high-flow conditions (open mouth). The remaining stations (i.e. P1–P3, and P5) act as stormwater conduits during high-flow events, transforming into standing pools or drying up under low-flow conditions. As such, these stations serve only as flow-driven and intermittent sources of allochthonous inorganic nutrients to the estuary.

Microalgal community dynamics

Phytoplankton concentrations were categorised *sensu* Lemley *et al.* 2015, by applying the 90th percentile, into 'oligotrophic' (0–≤5 μg Chl-*a* L^{-1}), 'mesotrophic' (>5–≤20 μg Chl-*a* L^{-1}), 'eutrophic' (>20–≤60 μg Chl-*a* L^{-1}) and 'hypereutrophic' (>60 μg Chl-*a* L^{-1}) conditions. Conditions remained oligotrophic during the closed (4.4 μg Chl-*a* L^{-1}) and open (1.97 μg Chl-*a* L^{-1}) mouth phases; however, once the mouth started closing and reached the 'semi-closed' state, conditions became eutrophic (23.3 μg Chl-*a* L^{-1}; Fig. 4). Phytoplankton bloom conditions (≥20 μg Chl-*a* L^{-1}) were recorded only after the artificial breaching event during the semi-closed mouth phase (Tables S2, S3). During the comprehensive surveys, the phytoplankton concentrations were spatially homogenous during the closed and open phases, and statistically different from those during the semi-closed phase ($\chi^2 = 27.22$, $P < 0.001$, $n = 58$; Fig. 4). Blooms during the semi-closed phase predominantly comprised *Cyclotella atomus* var. *marina* (Bacillariophyceae) in the

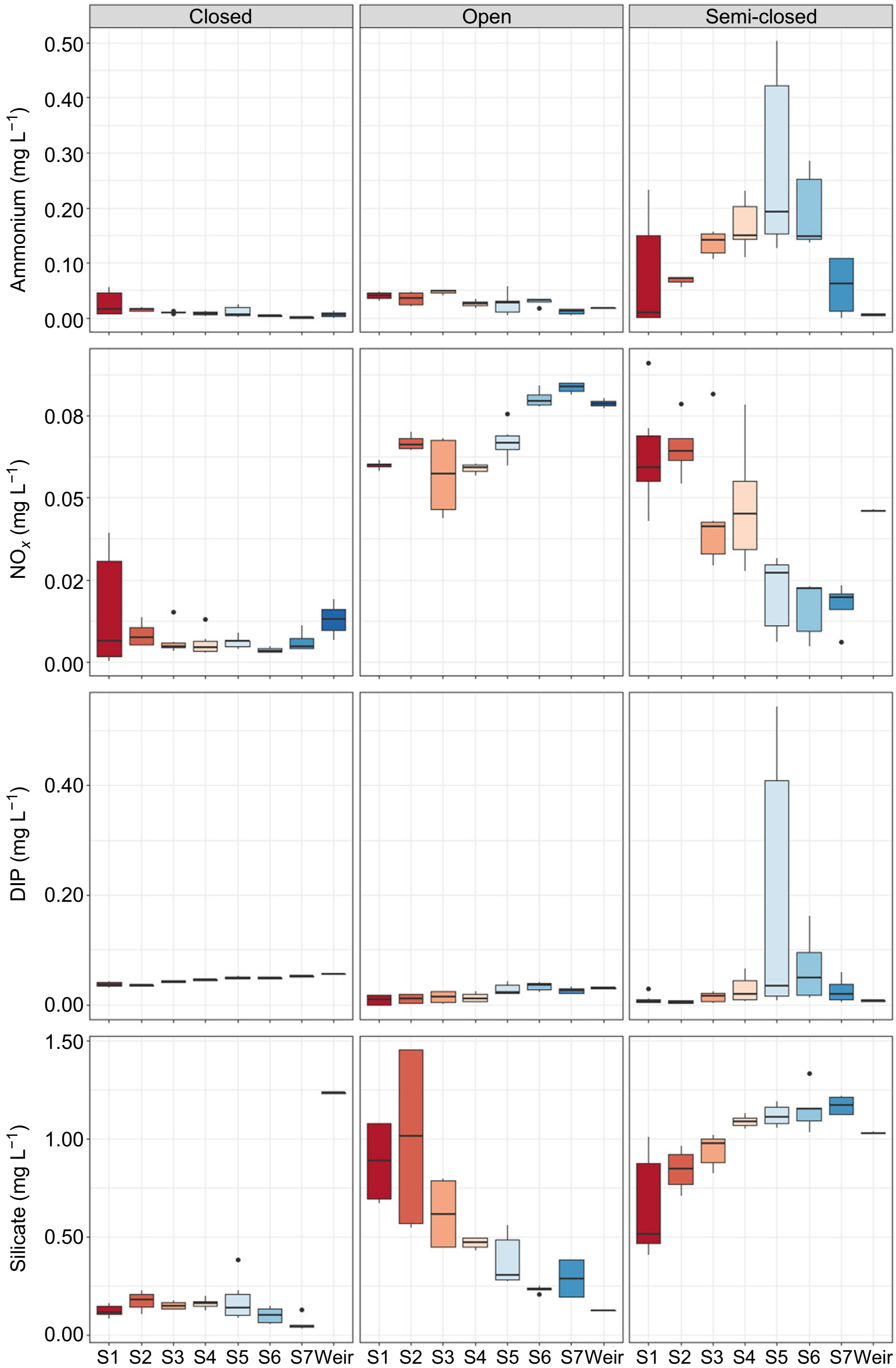

Fig. 6. Inorganic nutrient (NH_4^+, NO_x, DIP and dissolved silicate) concentrations (mg L^{-1}) for the comprehensive surveys (S1–S7 and Weir), i.e. closed (31 July 2021), open (26 November 2021) and semi-closed (18 February 2022) mouth states.

lower and middle reaches and *Cryptomonas* sp. (Cryptophyceae) in the upper reaches (S7; 6.0 km from the mouth; Table S2) of the estuary. The blooms occurred at the surface of the water column and were concentrated in the middle and upper reaches (Fig. 4). During the intensive sampling, phytoplankton biomass (μg Chl-*a* L^{-1}) and total cell abundance (TCA; log(TCA) + 1) both decreased after artificial breaching (Fig. 8). However, total cell abundance markedly increased after the breaching event, with both parameters reaching a peak during the semi-closed phase. Phytoplankton blooms

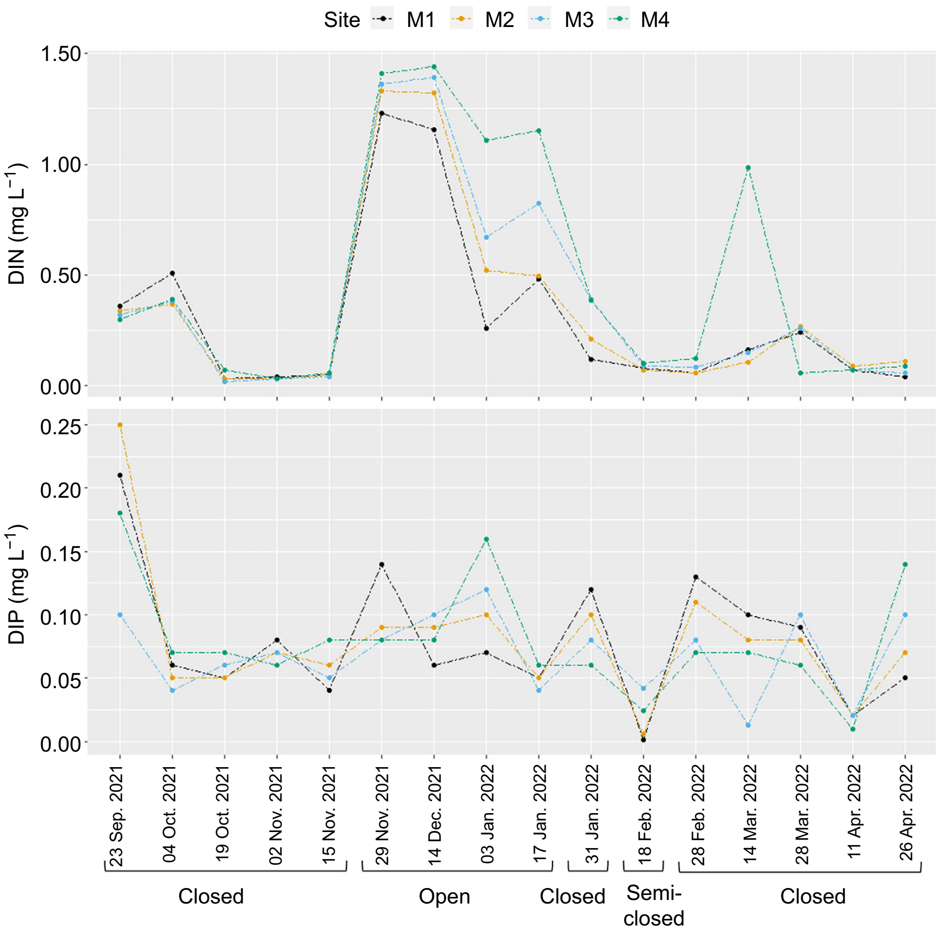

Fig. 7. Dissolved inorganic nitrogen (DIN) and phosphorus (DIP) concentrations (mg L^{-1}) during the intensive sampling program (M1–M4). The mouth state corresponding to each time period is displayed on the x-axis.

were observed only during closed (before and after breaching) and semi-closed mouth conditions. Four phytoplankton classes comprised these blooms (Table S3), namely Bacillariophyceae (*C. atomus* var. *marina*) during the semi-closed phase, and mixed blooms of Chlorodendrophyceae (*Tetraselmis* sp.), Cryptophyceae (*Teleaulax* cf. *amphioxeia*) and Dinophyceae (e.g. *Prorocentrum cordatum*, *Peridinium* sp.) during closed mouth conditions.

Microphytobenthos biomass is considered to be excessive at concentrations exceeding 100 mg Chl-*a* m^{-2} (Lemley *et al.* 2015). However, MPB biomass was below this threshold throughout the comprehensive surveys (Fig. 9). During the closed and semi-closed phase, MPB biomass decreased in concentration from the lower reaches to the upper reaches, whereas a spatial pattern was largely absent during the fluvially dominated open phase. Microphytobenthos biomass was different between mouth states ($\chi^2 = 12.91$, $P < 0.01$, $n = 63$), with concentrations during the closed state being markedly higher than during the open mouth state ($Z = 3.55$, $P < 0.01$). The Chl-*a* concentrations did not correlate significantly with the total sediment organic matter (Fig. 9), but comparisons of the means showed that the TOM in the open phase was also significantly lower than that in the closed phase ($Z = 2.77$, $P < 0.05$). The highest TOM values occurred in the semi-closed state, particularly at Site 6 (9.5 ± 0.9%), where they were markedly higher than any other value.

Discussion

The abiotic processes in the Great Brak Estuary have been well-monitored for the past 30 years through an adaptive management program (Van Niekerk *et al.* 2020*b*; Stein *et al.* 2021). Adaptive management aims to reduce the uncertainty of decision-making in environmental management through long-term monitoring and stakeholder engagement. Considerable efforts to improve biological status have been made, as seen by the eradication of the invasive *Spartina alterniflora* salt marsh grass (Adams *et al.* 2016). However,

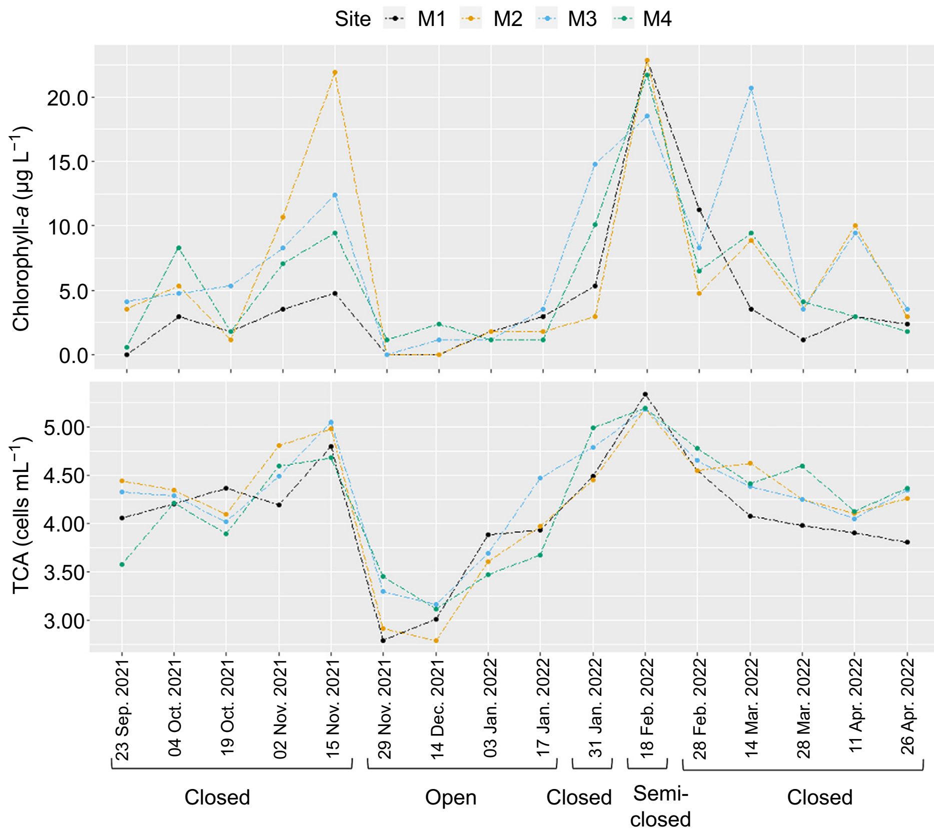

Fig. 8. Phytoplankton biomass (µg Chl-*a* L^{-1}) and total cell abundance (log(cells mL^{-1}) + 1) during the intensive sampling program (M1–M4). The mouth state corresponding to each time period is displayed on the *x*-axis.

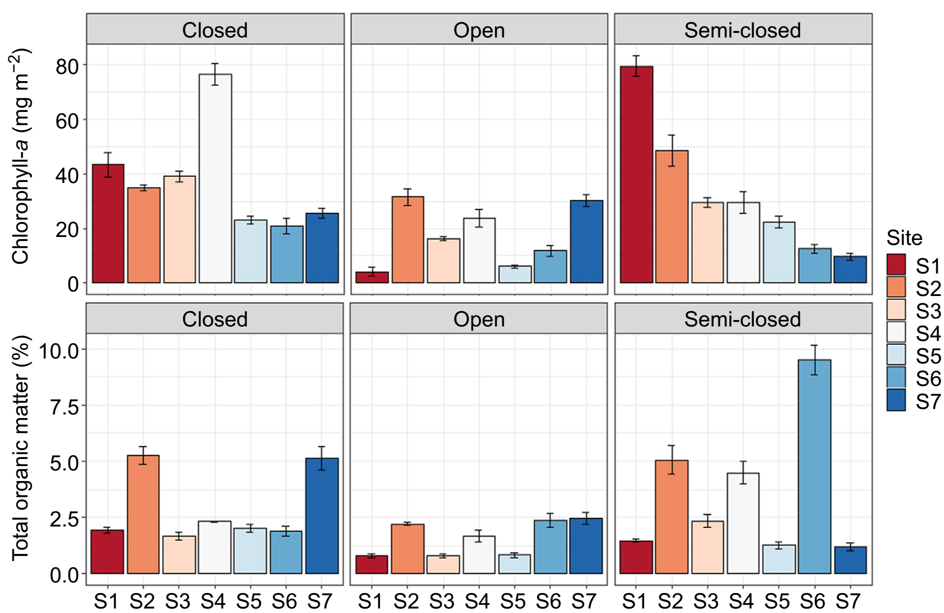

Fig. 9. Microphytobenthos biomass (mg Chl-*a* m^{-2}) and total sediment organic matter (%) during the comprehensive surveys (S1–S7 and Weir), i.e. closed (31 July 2021), open (26 November 2021) and semi-closed (18 February 2022) mouth states.

the recent trend of ineffective artificial breaching events has highlighted that the estuary receives inadequate environmental flows to maintain ecological health and functioning. As such, the objective of this study was to assess whether the ecological health of the estuary, on the basis of primary eutrophication indicators (Lemley *et al.* 2015), has declined in recent years as a result of altered hydrology and failed breaching attempts arising from ongoing drought conditions and freshwater abstraction. Overall, results indicated that prolonged periods of mouth closure and below-average annual inflows have increased the prevalence of eutrophic conditions in the estuary, with the frequency and magnitude of macroalgal and phytoplankton blooms increasing. Moreover, the continuous degradation of the estuary has increased the spatial extent of hypoxia to much shallower areas than previously reported. Thus, the proposed hypothesis was rejected, because despite maintaining open mouth conditions for nearly 3 months, the artificial breaching event documented in this study was unable to prevent eutrophic conditions. These trends are concerning and illustrate that the estuary is on a negative trajectory, potentially towards an ecological tipping point. Thus, the goal to improve the health of the Great Brak Estuary by 2025 will not be achieved with the current trajectory in environmental flows.

Mouth state

Estuaries require adequate environmental flows to maintain their ecological health (Adams and Van Niekerk 2020). The quantity and frequency of freshwater inflows to the Great Brak Estuary have decreased significantly since the construction of the Wolwedans Dam in 1989 (Human *et al.* 2015). Additionally, flood peaks have decreased by 84% in the past 2 years of the studied period, and median flows are five times lower than the average. A previous study indicated that prolonged mouth closure occurred in 2009–2010 because of inadequate environmental flow releases from the upstream dam and resulted in a loss of marine connectivity for 2 years (Nunes and Adams 2014). Since 2016, freshwater inflow to the estuary again decreased substantially because of ongoing drought conditions and continued abstraction. The cumulative effects of these hydrological shifts have resulted in the failure of recent artificial breaching attempts in 2020 and 2021, with mouth closure often occurring within 7 days of breaching instead of the prescribed minimum 1-month period under drought conditions (NCC Environmental Services and Anchor Environmental Consultants, unpubl. data). The initial breaching attempt during this study failed (i.e. closed within 4 days), yet flood conditions and subsequent additional water releases from the Wolwedans Dam ensured that marine connectivity was, albeit intermittently, maintained for nearly 3 months. Spillage volume for the period 22 November 2021 to 6 January 2022 is estimated at 15.9×10^6 m^3, which is similar in volume to the recommended annual environmental flow allocation to the system. Much of this flow was delivered as two large flood pulses (Fig. 2). Unfortunately, owing to low dam levels preceding the events, much of the volume and flushing potential of the initial flood pulse was attenuated by the dam. Additionally, significant accumulation of marine sediment over the past decade in the mouth region has resulted in the estuary becoming constricted less than a month after breaching. The sedimentation patterns in the estuary can also be influenced by increased sediment loads introduced from fluvial sources during high rainfall periods (Nunes and Adams 2014). Although high-flow periods typically flush sediments from TCEs (Human *et al.* 2016), the increased sediment loads associated with fluvial inputs in the Great Brak Estuary may contribute to sediment accumulation during failed artificial breaching attempts. This constriction prompted additional neap tide environmental flow releases every 2 weeks to maintain marine connectivity. This study demonstrated that the combined effect of reduced flood pulses and freshwater releases from the Wolwedans Dam failed to mimic the resetting effect of natural flow regimes, and instead increased estuarine vulnerability to eutrophic conditions (Schallenberg *et al.* 2010; Hoeksema *et al.* 2018; Adams and Van Niekerk 2020; Van Niekerk *et al.* 2022*a*; Mayjor *et al.* 2023).

Salinity

The water column of the Great Brak Estuary demonstrated physical conditions expected during each mouth state (Snow and Taljaard 2007). The period of mouth closure prior to breaching was characterised by well-mixed mesohaline conditions, as well as low *in situ* inorganic nutrients and phytoplankton biomass. However, these seemingly oligotrophic and well-oxygenated conditions (as is the natural state of the estuary) were largely a function of the extensive growth and nutrient uptake of dense macroalgal blooms (*Cladophora glomerata*) present during the initial closed phase (Nunes and Adams 2014; Human *et al.* 2015; Kibble *et al.* 2023). Subsequent to the flood-induced artificial breaching event, the estuary was characterised by oligohaline (0.5–5) conditions throughout. Vertical and horizontal salinity gradients re-established once freshwater inflow volumes subsided and the mouth began constricting (i.e. semi-closed state). Thereafter, managed post-flood releases were successful in resetting salinity gradients and recreating connectivity, but failed to scour enough sediment to prevent imminent mouth closure and well-mixed brackish conditions (because of wind-mixing) once freshwater releases were halted. This succession from stratified open mouth conditions (oligo- to euhaline) to well-mixed closed mouth conditions (meso- to polyhaline) align with those previously described for the estuary (Human *et al.* 2016) and is typical of small TCEs (Snow and Taljaard 2007).

Dissolved oxygen

The Great Brak Estuary was a well-oxygenated system in its natural state (Department of Water Affairs 2009). During this study, much of the water column remained well-oxygenated throughout the sampling period, regardless of mouth state. However, biologically stressful DO conditions (<5 mg L^{-1}) occurred in the deepest areas of the middle reaches. Similar observations were made by Human *et al.* (2016), with well-oxygenated conditions predominantly residing during closed- and open-mouth phases, with hypoxic conditions occurring only in the deeper middle reaches. This study indicated that the hypoxic zone occurs as a result of the change in flushing mechanisms after the construction of the Wolwedans Dam. As such, oxygen-deprived conditions are expected during closed mouth conditions in TCEs and can worsen during prolonged periods of closure (Nunes and Adams 2014; Human *et al.* 2016; Hoeksema *et al.* 2018). Decomposing organic matter from large macroalgal blooms during the closed mouth phase in the Great Brak Estuary (Kibble *et al.* 2023) can cause the hypoxia to spread as dead organic material sinks to the bottom sediments. The provision of sufficient environmental flows to estuaries has been shown to flush estuaries and help moderate hypoxia (Kurup and Hamilton 2002; Mayjor *et al.* 2023). A previous study by Human *et al.* (2016) highlighted the benefits of natural breaching events in promoting strong tidal exchange, sediment scouring and prevention of eutrophic symptoms (e.g. hypoxia, macroalgal accumulations) in the Great Brak Estuary. However, notably, the hypoxic zone in this study was not limited to just the deepest section of the estuary and, instead, was present throughout the shallower (~1 m) middle and upper reaches during the semi-closed mouth state subsequent to high flow conditions and artificial breaching. The persistent nature and spread of this hypoxic zone indicate that the drought conditions, continued abstraction, longer water residence times and preceding failed breaching attempts hindered the scouring potential of the high-flow event. As such, the current artificial breaching and freshwater dam release practices do not simulate the scouring effect of natural floods in the system; therefore, sediment build-up and anoxia are inevitable in the estuary.

Inorganic nutrients

Allochthonous nutrient loading to the Great Brak Estuary primarily originates from river inputs, as well as numerous peripheral flow-dependent point sources (Table S1). A study by Lemley *et al.* (2014*b*) suggested that catchment land-use practices are increasing nutrient concentrations and that the upstream dam may be acting as a sink that supplies the estuary with nutrients during high-flow events. However, annual DIN (4.7 kg day^{-1}) and DIP (1.2 kg day^{-1}) fluvial loads are typically low (Lemley *et al.* 2014*b*), accounting for less than 3% of the N and P budget of the estuary (Human *et al.* 2015). The largely autochthonous nature of nutrients in the system is evidenced by the sediment, submerged macrophytes and macroalgae serving as the major pathways for nutrient efflux (Lemley *et al.* 2014*a*; Human *et al.* 2015). As such, failed breaching attempts and reduced environmental flows reinforce negative feedback loops (i.e. internal nutrient loading) that act to maintain and promote eutrophic conditions over prolonged periods. The degraded state of the estuary was evidenced in this study by the initial widespread presence of macroalgal blooms during the closed state. These dense macroalgal accumulations incorporate excess nutrients and create low ambient nutrient conditions in the water column (Nunes and Adams 2014; Lemley *et al.* 2015). Extended periods of mouth closure can cause the local extinction of important macrophyte habitats, as seen in some TCEs in Australia (Scanes *et al.* 2020). It has been reported that extreme climatic events, such as the droughts experienced in the Great Brak Estuary, are the cause of such extinctions and that macrophyte recruitment may take 6–7 years to recover, given adequate water quality. Kibble *et al.* (2023) showed a significant decrease in submerged macrophyte (*Zostera capensis* and *Triglochin* spp.) abundance compared with historic results, with smothering by dense macroalgal mats an important driving factor thereof (Nunes and Adams 2014). During the open and semi-closed mouth phases that followed the initial closed state, inorganic nutrients were fluvially derived from upstream dam releases or spillages and caused moderate nutrient accumulation in the hypoxic depths. During anoxic conditions, release of bioavailable forms of phosphorus (orthophosphate) and nitrogen (ammonium) occurs, which leads to rapid uptake and growth of phytoplankton, microphytobenthos and macroalgae (Bonaglia *et al.* 2014; Human *et al.* 2015). The dominant primary producers are largely dependent on hydrological conditions, with macroalgae, rather than phytoplankton or benthic microalgae, tending to thrive in the Great Brak Estuary as a result of the frequency of closed mouth conditions. Artificial breaching provides a means of improving the water quality of the estuary by flushing nutrients out of the estuary; yet, data from this study indicated that an influx of nutrients occurred instead.

Microalgal community dynamics

Microalgal communities shift rapidly in response to nutrient availability and are, therefore, a better indicator of eutrophic conditions than are *in situ* nutrient concentrations alone (Lemley *et al.* 2017; Lemley and Adams 2020). For example, the relationship between inorganic nutrient loading and microphytobenthos is confounded by the fact that autochthonous nutrient sources are often the factor driving biomass accumulation (Nozais *et al.* 2001). During this study, microphytobenthos biomass remained below the threshold for concern (>100 mg Chl-*a* m^{-2}), with the highest concentrations occurring in the middle reaches during the closed phase (76.3 mg Chl-*a* m^{-2}), and the upper reaches during the semi-closed phase (79.4 mg Chl-*a* m^{-2}). Microphytobenthos generally proliferates under prevailing conditions in the closed phase, that is, stable sediment, salinity and current speed, all of which can result in high rates of remineralisation (Adams and Bate 1999; Nozais *et al.* 2001; Perissinotto *et al.* 2002). That said, however, the increased incidence of eutrophic symptoms in the Great Brak Estuary, arising from insufficient environmental flows, was highlighted by Lemley *et al.* (2015) who recorded high biomass accumulations (~110 mg Chl-*a* m^{-2}) despite open mouth conditions. The current study confirmed that microphytobenthos concentrations were highest during the closed and semi-closed phases, as per the conceptual understanding, and found no statistical relationship between nutrients and microphytobenthos biomass, but rather with salinity, because high biomass is associated with stable closed mouth phases.

Phytoplankton bloom conditions (>20 μg Chl-*a* L^{-1}) were observed during the closed and semi-closed mouth phases. At the onset of high-flow conditions, freshwater inflow from the upstream dam diluted phytoplankton abundance and introduced freshwater taxa, predominantly chlorophytes. Chlorophyte dominance typically indicates freshwater conditions and low residence times (Paerl *et al.* 2006; Barbosa *et al.* 2010; Kotsedi *et al.* 2012). Diatoms were dominant during the closed (max. ~1200 cells mL^{-1}) and semi-closed (max. ~20 000 cells mL^{-1}) mouth phases. This broad group of taxa has a wide salinity tolerance spectrum and is capable of accumulating in low nutrient concentrations (Ohrel and Register 2006). The dominance of small taxa such as *Cyclotella atomus* var. *marina* in this study during periods of prolonged water residency and well-mixed polyhaline conditions was expected, given that *in situ* nutrient availability is low due to excessive macroalgal growth (Nunes and Adams 2014; Lemley *et al.* 2015; Kibble *et al.* 2023). Phytoplankton communities tend to be dominated by small-celled taxa when nutrient concentrations are low, whereas larger-celled taxa are typically supported by increased nutrient availability (Cloern *et al.* 2016). The emergence of more motile, larger and competitive taxa, such as those belonging to Dinophyceae (e.g. *Prorocentrum cordatum*, *Peridinium quadridentatum*) and Cryptophyceae (e.g. *Teleaulax* cf. *amphioxeia*, *Cryptomonas* sp.), coincided with decomposition of macroalgal biomass (i.e. release of inorganic nutrients) and baseflow conditions (median ~0.02 m^3 s^{-1}; Table 1) that created favourable hydrodynamic conditions (e.g. reduced flushing potential). As such, phytoplankton blooms (≥20 μg Chl-*a* L^{-1}) in the Great Brak Estuary occurred just prior to, as well as after, the artificial breaching event, once freshwater inflow rates subsided (<0.01 m^3 s^{-1}), leading to increased water residency and the formation of haloclines. The blooms were limited to the surface waters and occurred in meso- to polyhaline waters throughout the estuary. Notably, a hypereutrophic bloom accumulation (≥60 μg Chl-*a* L^{-1}) of *Cryptomonas* sp. (Cryptophyceae) occurred in the vertically stratified upper reaches during semi-closed mouth conditions, being the largest bloom event ever recorded in the estuary. Additionally, the emergence of *P. cordatum* (formerly *P. minimum*) in the estuary is a concern given its capabilities to form toxic harmful algal blooms. These observations have provided further evidence of the inefficiency of current practices aimed at mimicking natural breaching events (e.g. timing, volume). As such, effective dam releases and mouth breaching practices are necessary to create a sudden ecological shift that (1) alleviates prolonged periods of macroalgal-dominance and (2) facilitates a more diverse primary producer community that includes microalgae (Larned 1998; Human *et al.* 2015; Lemley *et al.* 2018).

Implications for management

Artificial breaching is implemented in the Great Brak Estuary on an annual basis to maintain the biotic diversity and ecological health of the estuary (Van Niekerk *et al.* 2018, 2022*a*). The estuary has a rich history, entailing 30 years of monitoring and adaptive management practices that refined flow requirements to ensure environmental health; the first of its kind in South Africa (Van Niekerk *et al.* 2020*b*). During the construction of the Wolwedans Dam in the 1980s, only 1×10^6 m^3 of water was set aside for environmental flow releases, but studies later showed that this volume is 7–10 times smaller than that needed to maintain the condition of the estuary (Van Niekerk *et al.* 2019*c*). The freshwater input requirements have been further exacerbated by the impacts of climate change, with the possibility of reduced flooding frequency into the future being of particular concern (e.g. extended periods of closure, reduced scouring). During the past decade, managed water releases from the upstream dam have been used to maintain open mouth conditions after artificial breaching events. However, these releases have been largely ineffective because of the extensive build-up of organic matter and marine sediment near the mouth. In the years following the completion of the Wolwedans Dam, a study by Slinger *et al.* (2005) indicated that although the abiotic characteristics of the estuary had changed, the biotic characteristics were well maintained. However, the prolonged loss of marine connectivity in 2011, and again in 2020–2021, shows that maintaining the health of the estuary has become difficult without adequate environmental flows because of the loss of variability, particularly in the face of global change. Thus, an annual flow allocation (i.e. dam releases and spillages) of 11×10^6 m^3 has been recommended to prevent further deterioration in estuary health, as opposed to the historical allocation of 1×10^6 m^3 (Van Niekerk *et al.* 2020*b*). Once artificial breaching has been successfully implemented and a channel has developed, an additional flow release should be authorised to maximise sediment scouring (Van Niekerk *et al.* 2020*b*). Thereafter, depending on dam levels, open mouth conditions can be maintained over the spring–summer period by scheduling further water releases (i.e. 0.4 $m^3 s^{-1}$ for 4–6 days) during neap tides.

Because climate change is expected to increase the frequency and duration of droughts (Wetz and Yoskowitz 2013; Van Niekerk *et al.* 2022*b*), as well as marine wave action (storminess) in the Great Brak Estuary, it is important that the environmental flow requirements of the estuary are met to maintain adequate cycles of open or closed mouth phases. Management plans should be consistent with the desired outcome for the catchment (Stein *et al.* 2021), which is to improve the health of the estuary (Department of Water Affairs 2009). A higher allocation of freshwater from the upstream dam is imperative if the current decline in biological status is to be improved and an open mouth maintained. This, in turn, will also benefit local residents and the tourism industry, which are dependent on a healthy, functional estuary for their wellbeing and livelihood. However, the nutrient status of the freshwater releases and point-source inputs must be considered before such management actions are implemented. Restoration of the environmental health of the system should start now, before local extinction of important tidal-dependent macrophyte species that can extend recovery efforts by 6 or more years (Scanes *et al.* 2020).

Supplementary material

Supplementary material is available online.

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