Coorong water quality synthesis with a focus on the drivers of eutrophication

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Respect and Reconciliation

Aboriginal people are the First Peoples and Nations of South Australia. The Coorong, connected waters and surrounding lands have sustained unique First Nations cultures since time immemorial.

The Goyder Institute for Water Research acknowledges the range of First Nations' rights, interests and obligations for the Coorong and connected waterways and the cultural connections that exist between Ngarrindjeri Nations and First Nations of the South East peoples across the region and seeks to support their equitable engagement.

Aboriginal peoples' spiritual, social, cultural and economic practices come from their lands and waters, and they continue to maintain their cultural heritage, economies, languages and laws which are of ongoing importance.

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Executive summary

The Coorong is a Ramsar-listed estuarine lagoon system at the downstream end of the Murray-Darling Basin that has experienced declining ecological health over recent decades. This is likely due to a decrease in freshwater inflows from the River Murray, and the interactive effects of hypersalinity and eutrophication and changes in the water level regime. Eutrophication can be defined as an increase in the supply and accumulation of organic matter (e.g. algae) to an aquatic ecosystem and can cause many deleterious effects, including depletion of dissolved oxygen, toxicity (e.g. from harmful algal species, sulfide, ammonia), and loss of submerged aquatic vegetation and benthic ecosystems.

As part of the South Australian Government's Project Coorong initiative, the Healthy Coorong Healthy Basin (HCHB) program aims to achieve long term health of the Coorong by providing evidence-based solutions to both immediate threats and future conditions anticipated under a changing climate. One component of the HCHB Trials and Investigations Project is *Understanding Nutrient Dynamics*.

This report provides a synthesis of the long-term (>20 year) Coorong water quality data set aimed at assessing: (a) changes in Coorong water quality, with a particular focus on salinity, nutrients and eutrophication; and (b) the drivers for these changes. The findings for the Coorong were also compared to those in relevant international scientific literature.

Key findings were:

- Estuaries and lagoons in arid and semi-arid catchments such as the Coorong are particularly susceptible to hypersalinity and eutrophication following climatic and/or hydrodynamic changes that lead to reductions in flushing and increased water residence time.
- Based on analysis of the salinity, chlorophyll *a*, total nitrogen (TN) and phosphorus (TP) concentrations, the southern parts of the Coorong are persistently hypersaline and hypereutrophic. This was exacerbated during the extreme Millennium Drought period when no river inflows occurred, and salinity peaked near 200 mS/cm (approximately 5 times seawater salinity) and very high total nutrient levels were present.
- The results indicate that changed hydrodynamics and evapo-concentration processes in the Coorong has reduced flushing, especially in the South Coorong lagoon during the Millennium Drought. This has resulted in prolonged periods of extreme hypersalinity (EC > 120 mS/cm or salinity > 100 psu) and hypereutrophication (i.e. TP >0.2 mg/L, TN >4 mg/L, chlorophyll *a* >50 μg/L). In contrast, dissolved inorganic nutrients are generally low which, given the high chlorophyll *a* levels, suggests rapid recycling and uptake of any bioavailable nutrients by algae (e.g. phytoplankton and filamentous algae).
- Some phytoplankton die-off (based on chlorophyll *a* concentrations decreasing) appears to occur at the extreme end of the salinity range (EC > 120 mS/cm or salinity > 100 psu), likely due to salt toxicity. However, it is important to note that the data do not show a large drop in TN and TP levels and so eutrophication is persistent with the extreme hypersalinity, despite lowering chlorophyll *a* levels.
- The internal sources and cycling of nutrients within the system are currently quite unclear but specific process based research is underway as part of the HCHB Trials and Investigations *Understanding Nutrient Dynamics* component.
- There is some evidence of seasonally-higher availability of ammonium (50-100 μg/L as NH_4-N), which could be linked to the build-up and breakdown of organic matter/algae produced with the system, enhanced nutrient release from reducing sulfide-rich sediments, and/or inhibition of coupled nitrification-denitrification reactions.
- Phosphorus (P) appears to be the limiting nutrient for phytoplankton growth (i.e. the availability of P currently controls the amount of phytoplankton growth but this apparent effect could be driven by an oversupply of nitrogen).

We also hypothesise that the persistent hypersalinity in the South lagoon now reinforces the eutrophication process by negatively impacting benthic macroinvertebrates (nearly completely absent now) and seagrass (*Ruppia* sp.) communities, which would otherwise promote nutrient sequestration and elimination processes.

Based on the knowledge available, increasing system flushing (frequency and magnitude), in particular for the South Lagoon, to try and reverse eutrophication would at a conceptual level be beneficial to: (a) export nutrients, algae and organic matter; (b) reduce algae and total nutrient concentrations in the water column to reduce deposition of organic matter and nutrients to the sediment; (c) reduce algal-derived turbidity to enable increased light penetration for seagrasses (*Ruppia* sp.); (d) reduce hypersalinisation to enable re-establishment of benthic macroinvertebrates, and; (e) reduce formation of hypersaline, reduced, sulfide-rich sediments that we hypothesise are inhibiting healthy nutrient cycling.

The options to achieve increased system flushing include increased River Murray and South East catchment (via Salt Creek) inflows, and/or enhanced seawater inflows and connectivity. Many different options that could help achieve this are being investigated as part of the Coorong Infrastructure Investigations project of the HCHB program. Localised management actions such as *Ruppia* and benthic macroinvertebrate restoration could also help reduce eutrophication, particularly if system-wide water quality can be improved by increased flushing and reduced nutrient loadings. The *Understanding Nutrient Dynamics* component that this synthesis is part of will provide critical information to the feasibility, risks and benefits of various management options.

Acknowledgments

This project is part of the Department for Environment and Water's Healthy Coorong Healthy Basin Program, which is jointly funded by the Australian and South Australian Governments.

We would like to acknowledge Nikki Grigg of CSIRO, Jody O'Connor and Mark de Jong of DEW for their reviews that have improved this work.

This report is dedicated to our colleague Professor Peter Teasdale who sadly passed away during the finalisation of this report. Peter was a great person and scientist who we miss dearly.

1 Introduction

Eutrophication can be defined as an increase in the supply of organic matter to an ecosystem (Nixon 2009, Le Moal et al. 2019), and is a major and ongoing concern in aquatic systems worldwide (Cloern 2001, McDowell et al. 2020). The accumulation of organic matter can cause many deleterious effects including depletion of dissolved oxygen, toxicity (e.g. from harmful algal species, sulfide, ammonia), and loss of submerged aquatic vegetation such as seagrasses and benthic ecosystems (McGlathery et al. 2007, Nixon 2009). Two of the key drivers of eutrophication in estuaries have been identified as increased external nutrient inputs and/or reduced flushing (Bricker et al. 1999, Swaney et al. 2008, Steward and Lowe 2010, Le Moal et al, 2019), both of which can serve to increase organic matter supply. Elevated external nutrient loadings may occur where land use changes impact river catchments (e.g. increase in intensive agriculture or urbanisation) and lead to increased point and diffuse source pollution loads. However, external nutrient loading alone is not necessarily an accurate predictor of eutrophication (Bricker et al. 1999, Cloern 2001, Nixon 2009).

Reduced flushing of estuaries, which increases the residence time and retention of nutrients (and major ions, and thus salinity), commonly occurs when river or seawater inflows are altered via anthropogenic influences such as water extraction, construction of regulating structures, or climate change (Nixon 2009). The relative importance of evaporation may increase with prolonged water residence time, particularly in arid to semi-arid environments. A consequence of longer water residence times is an increase in the relative importance of various internal biogeochemical processes, which can result in enhanced recycling of bio-available nutrients under eutrophic conditions. Tight coupling of organic matter and nutrient cycling exists between pelagic and benthic compartments in shallow estuaries.

In shallow oligotrophic (low nutrient) systems, primary production tends to be dominated by benthic producers (e.g. benthic microalgae and seagrass), resulting in good sediment oxygen supply and uptake of bioavailable nutrients from the water column. A significant fraction of nutrients are retained in benthic biomass, and coupled nitrification (oxidation of ammonia to nitrite and nitrate)-denitrification (conversion of nitrate to nitrogen gas which can be lost from system) processes (microbially-mediated) can result in a significant sink of bio-available nitrogen (Seitzinger 1988). This balance is fundamentally shifted by an increase in the supply of bio-available nutrients from point and diffuse sources which favours the growth of phytoplankton and macroalgae (le Fur et al. 2009, Pérez-Ruzafa et al. 2019) and reduces water clarity. These changes typically lead to light limitation of benthic production, and the enhanced deposition of phytoplankton and macroalgal detritus leads to the organic enrichment of sediments. Microbially-mediated mineralisation (breakdown) of this organic matter rapidly depletes oxygen at the sediment surface, leading to a shift away from aerobic respiration to a predominance of anaerobic respiration pathways such as sulfate reduction. Reoxidation of reduced by-products is limited by oxygen supply, resulting in a build-up of sulfide phases which can be toxic to infauna and rooted macrophytes (Heijs et al. 2000). In addition, coupled nitrification-denitrification is inhibited by sediment anoxia, and the relative importance of dissimilatory nitrate reduction to ammonium (DNRA) increases (Hardison et al. 2015, Nizzoli et al. 2006). The net result of reduced benthic production and shifts towards anaerobic metabolism is enhanced recycling of bio-available nutrients to the water column and greater retention of nitrogen within the system (Kemp et al. 1990). This in turn promotes further growth of phytoplankton and macroalgae at the expense of seagrasses.

Lagoons and estuaries in arid-semi-arid climates such as the Coorong have received much less attention than their temperate counterparts (Cloern 2001). Due to their generally higher evaporation rates and lower freshwater inflows, these systems could be at higher risk from hydrological and climatic shifts that reduce flushing due to irrigation water extractions and climate change. In estuarine-lagoon systems with low river inputs, hypersalinisation could exacerbate eutrophication as organic matter gets concentrated along with salt, as residence times increase (Bricker et al. 1999, Cloern 2001). Micro-tidal estuaries and lagoons (those with restricted tidal exchange for substantial periods) are also inherently vulnerable to hypersalinisation (Warwick et al. 2018, Tweedley et al. 2016).

The Coorong micro-tidal lagoon represents the terminus of Australia's largest river catchment, the arid to semi-arid Murray-Darling Basin. The state of the Ramsar-listed Coorong ecosystem has declined significantly over recent years with reductions in waterbirds (Paton et al. 2009), aquatic plants (Dick et al. 2011, Kim et al. 2013), and macro-invertebrates (Dittmann et al. 2015), as well as a proliferation of filamentous algae (Collier et al. 2017, Brookes et al. 2018). The Coorong has experienced increased frequency and severity of hypersaline conditions that have contributed to the ecosystem decline (Gibbs et al. 2018). Loss of the associated ecological functions may also have profound influences on nutrient cycling. For example, rooted aquatic plants, such as *Ruppia* sp., remove nutrients from the water and sediment and oxygenate the sediment, reducing sulfide build up and phosphorus fluxes (Heijs et al. 2000, Pagès et al. 2012). Burrowing and bioturbating macroinvertebrates also oxygenate the sediment, encouraging the breakdown of organic matter and the formation of Fe(III) oxy(hydroxides) that bind and sequester phosphate (Di Toro 2001), as well as promoting nitrogen elimination via nitrification and denitrification (Welsh 2000, Stief 2013). Hypersalinity over approximately 60-65 practical salinity units (psu) generally results in near complete loss of macroinvertebrates (Dittmann et al. 2015, Remailli et al. 2018), and their associated nutrient and organic matter processing functions.

We evaluated whether reduced flushing in a hypersaline lagoon aggravates eutrophication and leads to significant ecosystem decline by analysing long term datasets for the Coorong. Our synthesis of a long-term (>20 year) data set aimed to assess: (a) changes in Coorong water quality in relation to hypersalinity; and (b) the drivers for these changes. The findings are placed in the context of lagoons and micro-tidal estuaries globally, particularly those in Mediterranean climate regions and/or those susceptible to severe hydrological alterations.

2 Methods

2.1 Study area description

The Coorong is a shallow and narrow lagoon system, which runs north-west to south-east, parallel to the South Australian coast for ~110 km and separated from the sea by a sand barrier (Figure 1). The Coorong naturally splits about halfway along its length into the North and South Lagoons at a narrow constriction (near Parnka Point, Figure 1) that is approximately 100 m wide, and transitions from saline to hypersaline in the south. The average widths of the North and South Lagoons are 1.5 and 2.5 km respectively, and the average water depths are 1.2 and 1.4 m respectively (Gibbs et al. 2018). The Coorong has a constricted channel connection to the sea towards its north-western end, referred to as the Murray Mouth. Exchange of seawater occurs from the Murray Mouth into the main body of the Coorong North Lagoon, but the tidal flow is currently restricted. Water level variation in the Coorong is thus less driven by tides, but rather inflows from the Murray-Darling Basin and winds/storms. The River Murray enters into Lake Alexandrina and is regulated to flow, when sufficient water is available, out to the Murray Mouth-Coorong via barrage structures. The Tauwitchere Barrage is the largest and also closest barrage to the main body of the Coorong, and hence may exert the most influence on water quality under certain release conditions (Mosley et al. 2016).

The South Lagoon also receives inputs of fresh to brackish water from a network of drains from the South-East region of South Australia, discharging at Salt Creek. This discharge is generally seasonal and can be regulated via structures in Morella Basin immediately upstream from Salt Creek, and further upstream in the drainage network. Groundwater discharge zones exist along the length of the Coorong, from both perched freshwater lenses and the regional unconfined aquifer, but hydraulic gradients are relatively low (Haese et al. 2008, Barnett 2018).

Evaporation of the lagoon water concentrates major ions and other solutes in the water column, thereby inducing carbonate precipitation, particularly in the hypersaline regions (Shao et al. 2018). Mean maximum and minimum temperatures are 20.9 and 11.9°C respectively, while mean annual rainfall is 383 mm

(2003–2020 statistics, Bureau of Meteorology Hindmarsh Island Station, ID 023894). The site is exposed to regular coastal winds that cause mixing which, coupled with the lagoon's shallow nature, results in little salinity stratification except near the Murray Mouth during significant barrage releases (Geddes and Butler 1984).

2.2 Sample sites and methods

Water quality has been monitored by South Australian Government agencies since 1998 at twelve sites spanning the North and South Lagoons (see Figure 1, with the sample site coordinates listed in Supplementary Material S1). There are some gaps in the record where no monitoring occurred, most notably from mid-2011 to mid-2013 and mid-2016 to mid-2018. In the pre-2014 period, monitoring was conducted approximately monthly while post-2014, sampling events were conducted fortnightly over winter for a period of approximately 3–4 months (to coincide with the main period of inflows from Salt Creek) followed thereafter on a monthly basis. Water samples were collected via grab sampling from just below the water surface in polyethylene plastic bottles in accordance with standard methods (APHA 2005). Subsamples for dissolved nutrient analyses were immediately filtered through a coupled 1.2 µm syringe pre-filter (Sartorius GF) and 0.45 µm membrane filter (Sartorius Minisart). The syringe was rinsed with the sample and a small volume of water was passed through the filter prior to sample collection. Immediately following sampling, all samples were stored in the dark on ice, and transported to the laboratory within 12-24 hours.

2.3 Laboratory analytical methods and quality control

The water quality parameters analysed in the samples were specific electrical conductivity (EC, mS/cm at 25°C), pH, turbidity, nutrients (total nitrogen, TN; total Kjehldahl nitrogen, TKN; ammonium, NH_4; oxidised nitrogen, NO_x (nitrate + nitrite); total phosphorus, TP; filterable reactive phosphorus, FRP), chlorophyll *a*, and reactive silica. All analyses were undertaken at the Australian Water Quality Centre (AWQC), a National Association of Testing Authorities (NATA) accredited laboratory. EC was measured using a calibrated conductivity meter using high ionic strength standards. The relationship between EC and salinity is non-linear and problematic to define in hypersaline waters but can be approximated in the Coorong by (unpublished AWQC laboratory relationship):

$$\text{Salinity (psu)} = -7\text{E-}06 \times EC^3 + 0.003 \times EC^2 + 0.5865 \times EC \qquad \text{(Equation 1)}$$

where EC is in mS/cm units.

Chlorophyll *a* was measured spectrophotometrically following 95% ethanol extraction (APHA 2005). Nutrients were measured by standard colorimetric methods, specifically NH_4 by a phenate method, NO_x by the cadmium reduction method, FRP by an ascorbic acid reduction method, TKN by reaction with salicylate following digestion, reactive silica by the heteropoly blue method, and TP by the automated ascorbic acid reduction method (APHA 2005). TN was calculated from the sum of TKN and NO_x.

Quality control procedures are a routine and audited requirement of a NATA accredited laboratory, including routine analysis of blanks, replicates and certified reference materials. Since 1995 the AWQC has participated and demonstrated proficiency in an inter-laboratory comparison program providing natural pristine and impacted samples sourced from fresh, estuarine and seawater environments established by the Environmental Nutrient Collaborative Trial Committee.

2.4 Hydrology and meteorology

River Murray-Lower Lakes barrage outflow calculated data (based on gauging of the structures), including the split from the individual barrages, were obtained from the Department for Environment and Water (DEW), South Australia. Salt Creek gauging station (ID A2390568) discharge data were also obtained from DEW. Local meteorological data (wind speed, air temperature and rainfall) used to assess relations with water quality was obtained from the Australian Bureau of Meteorology Hindmarsh Island station (Site ID 23894).

2.5 Data analysis

The water quality data was downloaded from AWQC's database. The complete dataset (including additional water quality parameters and one-off sampled monitoring station data) and associated Python (v3.7) script is provided in an accompanying dataset.

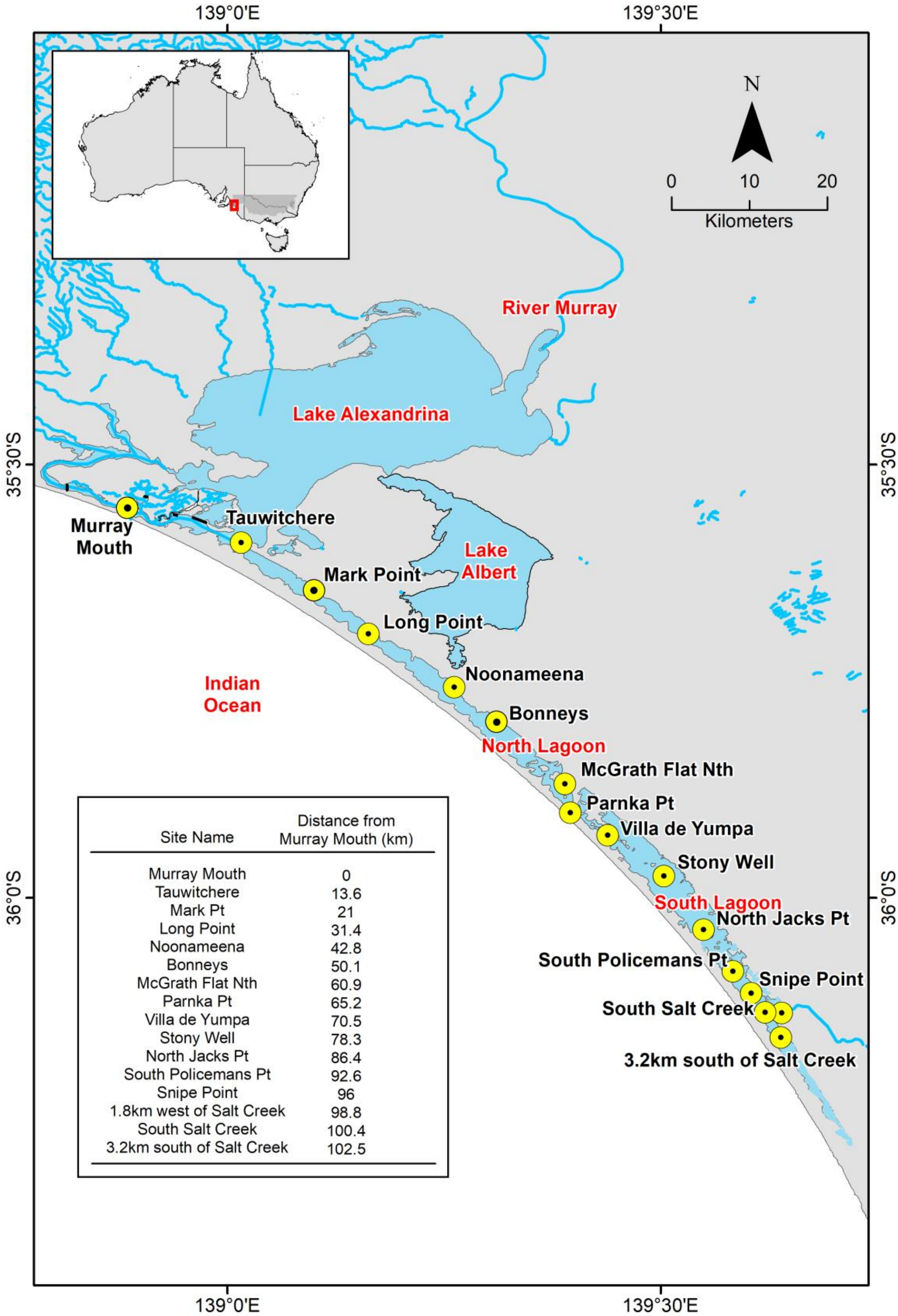
Using a locally weighted linear nonparametric regression method (LOESS), a quadratic function was fitted to the water quality versus salinity data. This was performed by using the "skmisc.loess" package in Python (v3.7) with a span of 0.75 to capture the trends within the data while avoiding overfitting.

A Trophic Index (TRIX) was calculated (Vollenweider et al. 1998, Giovanardi and Vollenweider 2004) to assess the degree of eutrophication in the Coorong:

$$\text{TRIX} = (\log10\ (\text{Chl}\ a\ x\ \text{DO\% deviation}\ x\ \text{TN}\ x\ \text{TP}) + 1.5)\ /\ 1.2 \qquad \text{(Equation 2)}$$

where Chl *a* is the chlorophyll *a* concentration as µg/L; DO% is dissolved oxygen as percent deviation from saturation, TN is the total nitrogen as µg/L and TP is the total phosphorus as µg/L. Measured values of Chl *a*, TN and TP in the Coorong were used in the equation along with a DO% of 50% (i.e. DO range of 50-150%) based on available field data. TRIX values greater than 6 are considered to represent eutrophic coastal waters.

The drivers of water quality change were explored through a Principal Components Analysis (PCA) on all the parameters in the hydrological, meteorological, and water quality dataset. PCA reduces a large number of variables to a few independent, composite variables (principal components) that attempt to explain much of the variance of the original data. Variables that are highly correlated with a principal component can indicate that they share a similar pattern or trend to that principal component, which are ordered hierarchically according to variance explained. The water quality data was normalised using the Kendall Tau rank procedure. The principal axis method was used to extract the components and this was followed by an oblique (oblimin) rotation. All data analysis procedures were performed using XLSTAT™. Factors were considered significant and were retained for rotation when eigenvalues were greater than 1 and they occurred prior to the change in slope of the associated scree plots.



Site Name	Distance from Murray Mouth (km)
Murray Mouth	0
Tauwitchere	13.6
Mark Pt	21
Long Point	31.4
Noonameena	42.8
Bonneys	50.1
McGrath Flat Nth	60.9
Parnka Pt	65.2
Villa de Yumpa	70.5
Stony Well	78.3
North Jacks Pt	86.4
South Policemans Pt	92.6
Snipe Point	96
1.8km west of Salt Creek	98.8
South Salt Creek	100.4
3.2km south of Salt Creek	102.5

Figure 1. Map of Coorong study area showing the location of monitoring stations. The inset table shows the distance from the Murray Mouth to the monitoring stations. The inset map shows Australia, along with the Murray-Darling Basin (grey-shaded area).

3 Results and discussion

3.1 Spatial variability in water quality

The spatial pattern of water quality from 1998-2020 showed strong gradients for most parameters along the Coorong from the Murray Mouth (Figure 2). From north to south, the EC increases towards Parnka Pt, which separates the North and South Lagoons (approximately 65 km from the Murray Mouth; vertical dashed lines in Figure 2). The EC is on average less than 50 mS/cm towards the Murray Mouth where estuarine conditions are predominant for approximately 20 km. On average, the South Lagoon has been extremely hypersaline (EC > 100 mS/cm which is approximately equivalent to a salinity of >80 psu for the recorded period). Slightly lower EC (EC ≈ 90 mS/cm) is present on average around the Salt Creek inflow (approximately 95 km from the Murray Mouth). The general north-south, estuarine-hypersaline, salinity gradient is generally consistent with that described previously under lower flow conditions (Geddes and Butler 1984).

Similar spatial patterns to EC are present for TP, TN and chlorophyll *a*, with higher concentrations in the South Lagoon grading to much lower concentrations in the North Lagoon. TP, TN and chlorophyll *a* concentrations in the South Lagoon are high relative to most other estuaries in southern Australia (Woodland et al. 2015) and globally (Smith et al. 2006). This is also reflected in the concentrations of these parameters exceeding the national water quality guideline trigger values (horizontal lines/zones on Figure 2 – South Central Australia Values, where available) (ANZG, 2018). The use of such guidelines is more uncertain for hypersaline water bodies that for fresh and estuarine water bodies . Nevertheless, the South Lagoon appears to be in a persistent hypereutrophic state currently. Notably, chlorophyll *a* and TP concentrations in the Coorong are persistently higher than those recorded earlier by Geddes and Butler (1984; generally <10 μg/L chlorophyll *a* and <0.1 mg/L TP).

Dissolved inorganic nutrient concentrations (NH_4, NO_x, FRP) in the South Lagoon show different spatial patterns to total nutrients (TN, TP), and concentrations are orders of magnitude lower (Figure 2). This indicates the total nutrient pool is mainly comprised of high concentrations of less biologically available organic nutrients (i.e. present in algae, microbes and detrital material in the water column). NO_x concentrations >0.01 mg/L are present near the River Murray-Lower Lakes flow inputs (e.g. near Tauwitchere barrage at 13.6 km from the Murray Mouth; Figure 2) but concentrations are in general low and often near or below detection limit (<0.005 mg/L) in the South Lagoon. Higher levels of dissolved nutrients are also evident immediately adjacent to the Salt Creek discharge into the South Lagoon (approximately 95 km from the Murray Mouth; Figure 2).

Ammonium often has higher concentrations (50-100 μg/L as NH_4-N) than the other dissolved nutrients (Figure 2) which may reflect the anoxic and sulfide-rich sediment conditions, resulting in the inhibition of nitrification (Kemp et al. 1990) and an increased NH_4^+ flux from the sediments (Di Toro 2001). Average NH_4^+ concentrations also exceeded guideline values at some sites (see horizontal lines on Figure 2). The Seagull Island to Stony Well area of the South Lagoon (78–93 km from the Murray Mouth) has higher NH_4^+ and FRP concentrations on average than other sites, while NO_x remains low. This also is consistent with a common dissolved nutrient source from anoxic sediment. While the dissolved inorganic N (DIN) component of the TN is low, the dissolved organic nitrogen (DON) fraction is unclear.

Turbidity is in general higher in the South Lagoon compared to the North Lagoon, and average levels greatly exceed national water quality guidelines in the South Lagoon (Figure 2). Lower turbidity is present in the South Lagoon near the Salt Creek inflow. In contrast higher turbidity is present on average near the Tauwitchere barrage (13.6 km from the Murray Mouth; Figure 2), likely reflecting high turbidity source water inputs from the River Murray-Lower Lakes (Mosley et al. 2012, Biswas and Mosley 2018).

3.2 Temporal variability in water quality

Time series (1998–2020) plots of water quality at selected sites (North Jacks Pt in the South Lagoon and Long Point in the North Lagoon) show high temporal variability (Figure 3). River Murray-Lower Lakes barrages and Salt Creek inflows are also displayed. The influence of a severe hydrological drought ("Millennium Drought"; 2002–2010), with extended periods of low to zero River Murray inflow conditions (see Mosley et al. 2012), is apparent in the data. The extreme lack of River Murray inflows during 2007–2010 appears to have had a major impact on lagoon salinity, which increased to peak at approximately 5 times seawater salinity (≈200 mS/cm; equation 1). This was due to reduced lagoonal flushing and salt export and the increased influence of evaporative concentration. Salinities in general are now higher on average compared to the 1960s-1980s (Gibbs et al. 2018; see Supplementary Material S5). In the mid-1970s, when high River Murray inflows were present, the South Lagoon approached marine salinities and the North Lagoon was estuarine (Geddes and Butler 1984). Declining river flows and high flow pulses have also resulted in sand build-up and channel constrictions near the Murray Mouth (Bourman et al. 2018), likely further reducing lagoonal flushing.

During the early period of the Millennium Drought (2002–2008), TN and TP became elevated in the South Lagoon, although at the extreme hypersaline conditions (EC > 110 μS/cm) in 2008–2010, concentrations appeared to stabilise or decline, as did chlorophyll *a*. This likely reflects reduced external nutrient inputs and/or also extreme hypersalinity impacts on primary productivity as discussed further below. Following the end of the Millennium Drought in late 2010, EC, TN and TP concentrations returned to pre-drought levels likely due to larger River Murray flows in 2011–2013 and late 2016, possibly assisted by increased Salt Creek discharges (post-2011; Figure 3). Turbidity was lower during the Millennium Drought which is likely due to reduced or zero turbid River Murray-Lower Lakes inputs, but has increased since then. pH is maintained at typical values for estuarine-marine systems (pH 7.8–8.5; see Supplementary Material S2).

Dissolved nutrient time series, plotted on log-scale in Figure 4, show a great deal of variability with many values at or near the laboratory analytical detection limit. In the extreme 2007–2010 period of the Millennium Drought, NH_4^+ and FRP appeared elevated in the South Lagoon (North Jacks Pt site) relative to the North Lagoon (Long Pt site). With higher River Murray-Lower Lakes inflows post-2011, higher levels of NH_4^+ and NO_x have often been present at Long Pt. Reactive Si concentrations generally follow similar patterns between the lagoons but slightly higher concentrations are present in the South Lagoon.

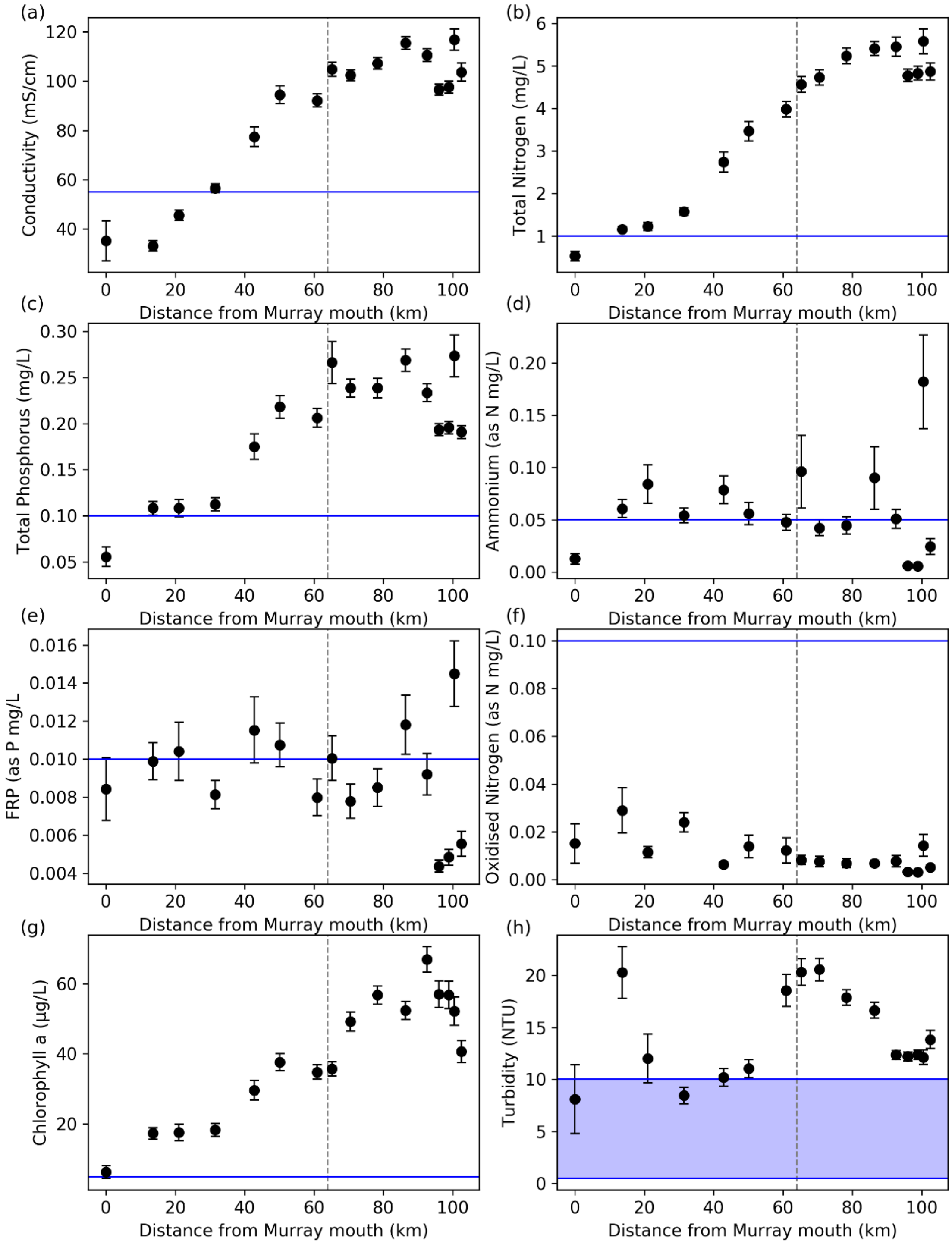


Figure 2. Box plots of median water quality values between 1998 and 2020 at monitoring sites in the Coorong, plotted vs distance from the Murray Mouth (see Fig. 1). Water quality parameters include electrical conductivity (Conductivity), total nitrogen, total phosphorus, ammonium, filterable reactive phosphorus (FRP), oxidised nitrogen, chlorophyll *a* and turbidity. The error bars represent standard errors of the mean value. Horizontal lines/zones show default guideline values, where available, from ANZG (2018).

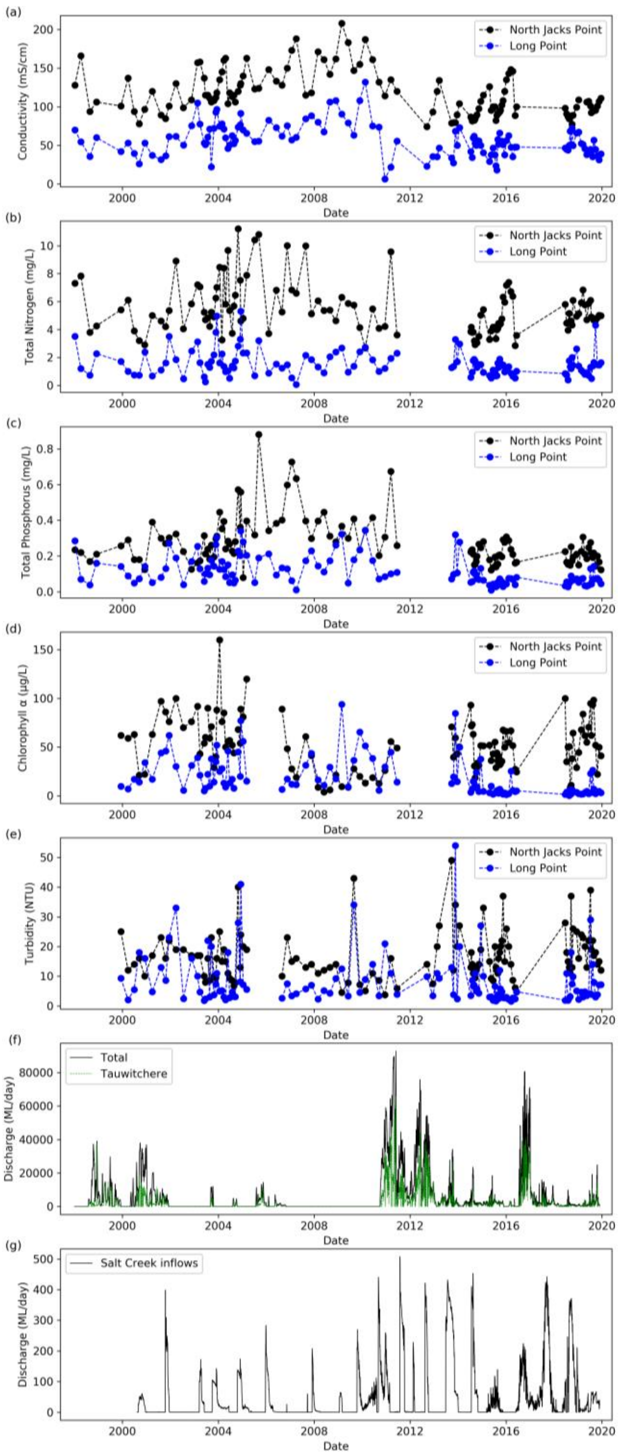


Figure 3. Time series (1998-2019) plots of water quality at North Jack Point (in South Lagoon) and Long Point (in North Lagoon) in the Coorong. Water quality parameters include electrical conductivity (conductivity), total nitrogen, total phosphorus, chlorophyll *a* and turbidity. Also shown are time series plots of inflows to the Coorong from (i) the Lower Lakes barrages into the North Lagoon (note the closest Tauwitchere Barrage data is also plotted separately in green as well as total barrage discharge in black), and (ii) Salt Creek discharge into the South Lagoon.

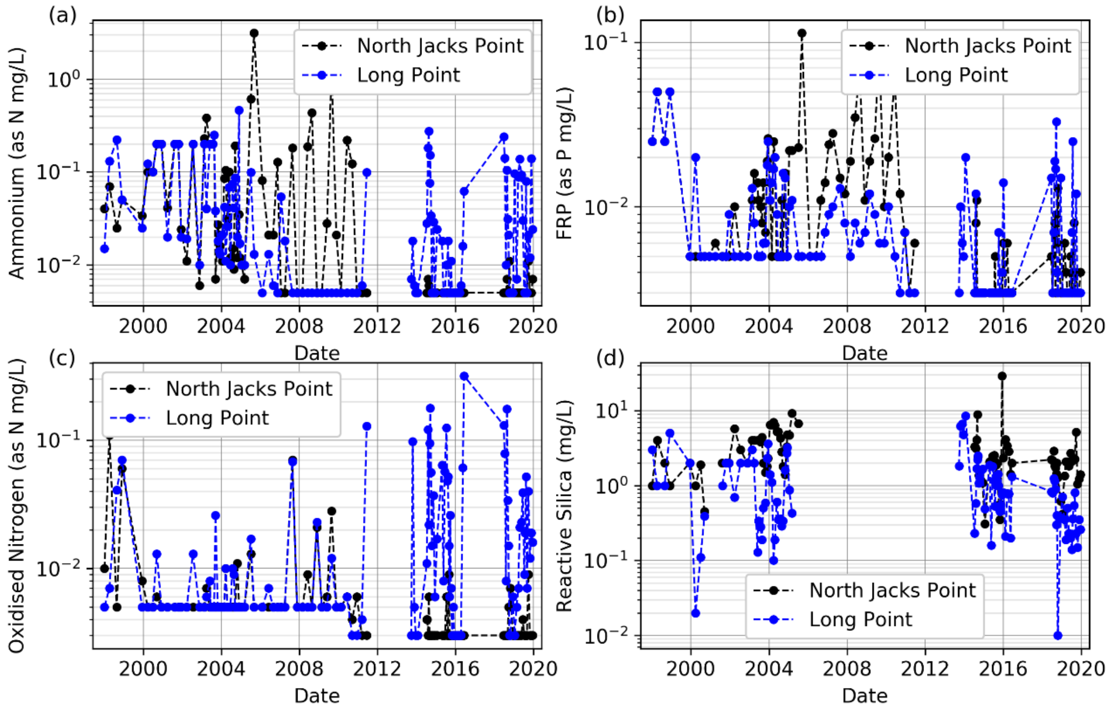


Figure 4. Time series (1999-2019) plots of dissolved nutrients at North Jack Point (South Lagoon) and Long Point (North Lagoon) in the Coorong. Dissolved nutrients shown include ammonium, filterable reactive phosphorus (FRP), oxidised nitrogen and reactive silica.

3.3 Drivers of eutrophication

To detect drivers for the dynamic changes in water quality in the Coorong over space and time, we assessed how flushing, dilution and evaporative concentration processes may change water quality parameters. TN, TP and chlorophyll *a* concentration, and TRIX (Trophic Index) showed decreasing trends from freshwater-estuarine conditions towards seawater salinity conditions (EC ≈ 55 mS/cm) (Figure 5). As EC increases past seawater values, these parameters also increased in a generally linear manner, although there is levelling off or decrease at extreme salinity, particularly for chlorophyll *a* (Fig. 5). The integrated eutrophication index TRIX shows similar patterns (Figure 5), with the observed values >8 in the hypersaline salinity range being consistent with highly eutrophic conditions (Vollenweider et al. 1998). Increased chlorophyll *a* concentration with increasing salinity has been observed at other sites globally where reduced or restricted flows and ocean inputs have occurred (Cloern et al. 2014). The declining chlorophyll effects we have observed at extreme salinity ranges appear to not have been observed previously. We hypothesise that phytoplankton die-off occurs at the extreme salinities (EC > 120 mS/cm or salinity > 100 psu) in the South Coorong, likely due to salt toxicity. However, it is important to note the data do not show a large drop in TN and TP levels and so eutrophication (i.e. organic matter supply; Nixon 2009) is persistent with the extreme hypersalinity, despite lowering chlorophyll *a* levels.

Results from the PCA correlating multiple water quality and predictor variables (hydrology, meteorology) are shown in Figure 6. The first three factors that were selected (following inspection of the point of inflection in the PCA Scree Plot; see Supplementary Material Table S3) describe 58% of the variability in the dataset. Several key eutrophication indicators (TN, TP, chlorophyll *a*) cluster together along with salinity but these are orthogonal (90%) to the hydrological variables (Figure 6). This suggests that while the relationship between total nutrients and salinity is strong, the barrage or Salt Creek inflow at the time of water quality sampling is less important. The squared cosines of given variables shown in Table 1 are in support of this with total nutrients (TN, TP), reactive silica and salinity loading on the first factor (F1). Bivariate Spearman Rank correlation coefficients also show moderate to strong relationships with salinity (see Supplementary Material

S4). We suggest the Coorong salinity and TN and TP are likely more influenced by: (i) River Murray inflow patterns over a longer time period (i.e. monthly-annual); (ii) the influence of marine water incursions through the Murray Mouth; (iii) changes in local water evaporation rates; and (iv) lagoon-channel connectivity that enables system flushing. We have demonstrated how hypereutrophication can develop and persist with an extreme lack of flushing, and this Coorong study outcomes are valuable in a global context where there are limited studies on estuaries in large arid/semi arid river catchments that are very susceptible to hydrological drought.

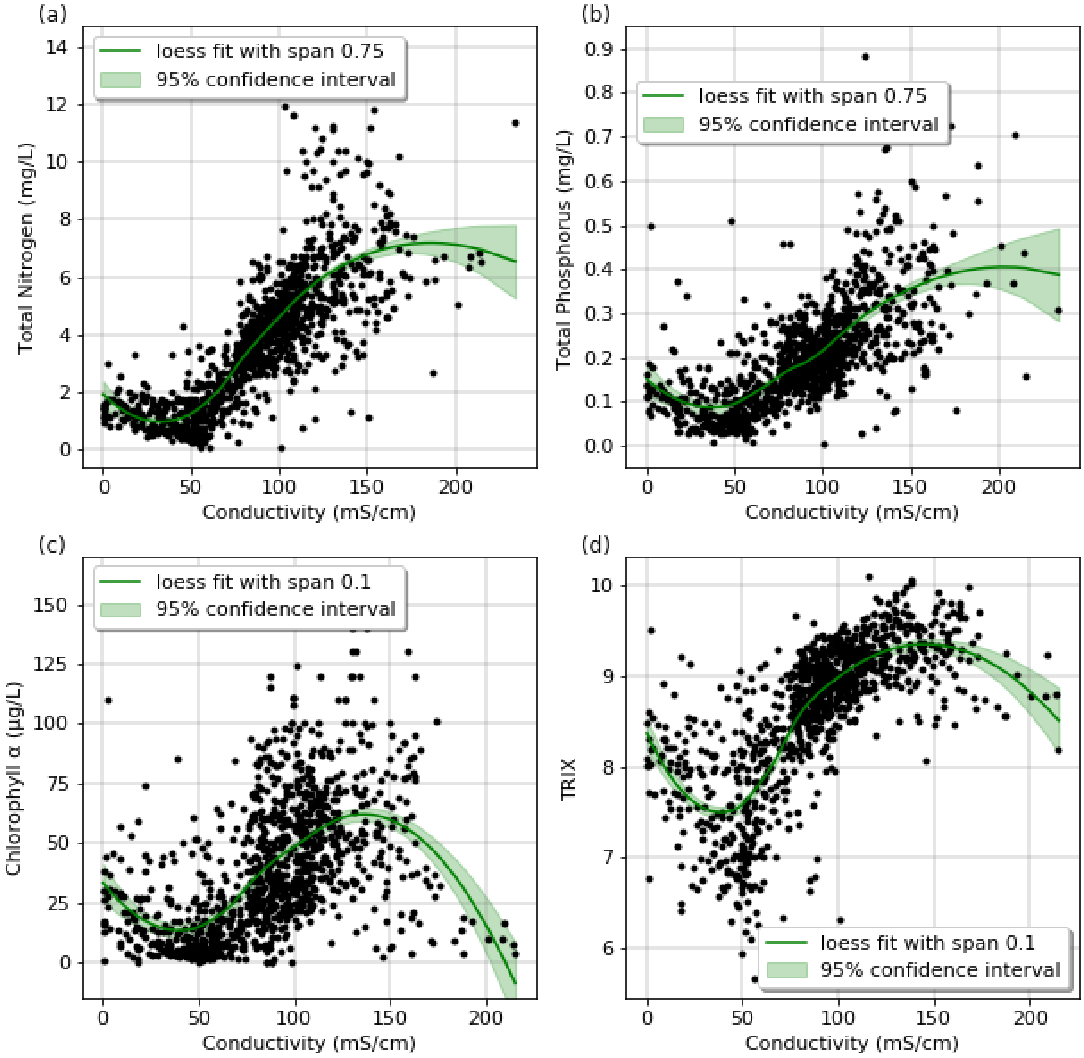


Figure 5. Relationship between water quality parameters and electrical conducitivity (conductivity) in the Coorong. Water quality paramters shown are total nitrogen, total phosphorus, Chlorophyll *a* and trophic index (TRIX).

Dissolved nutrients (NH_4^+, FRP, NO_X) plot on the opposite side of the PCA correlation circle to barrage flows (Figure 6), and loaded on the 2nd factor in the PCA, along with chlorophyll *a* (Table 1). This suggests that higher River Murray-Lower Lakes inflows via the barrages in general reduce dissolved nutrient and algae concentrations in the Coorong, likely due to enhanced dilution and lagoonal flushing during these periods (Grigg et al. 2009). The effect of Salt Creek inflows on overall Coorong water quality appears less pronounced based on this variable plotting towards the centre of the PCA correlation circle (Figure 6) with low squared cosine values (Table 1). The volume of these flows is lower relative to River Murray-Lower Lakes barrage inflows but directly discharges to the South Lagoon. Higher Salt Creek inflow periods have been noted to reduce TN and TP levels in the South Lagoon, which has been ascribed to driving export to the North Lagoon (Mosley et al. 2017). Meteorological variables load on the 3rd factor and wind speed appears more closely associated with FRP. This could indicate a role of wind in releasing phosphorus from the sediment pore water due to disturbance/resuspension (Di Toro 2001) but this requires further investigation and squared cosine values are low (Table 1).

There are additional processes affecting water quality that are not considered in the PCA, in particular internal nutrient cycling processes which have received limited research in the Coorong (Ford 2007, Grigg et al. 2009, Aldridge et al. 2018). The high algae and total organic nutrient loads in the lagoon water likely contribute to high organic loads and nutrient fluxes to and from the sediment. Indeed, the predominant organic matter composition in the Coorong surface sediment has been found to be from an algal derived source, with a larger contribution of seagrasses before Murray-Darling Basin water regulations and extractions were implemented (Krull et al. 2009; McKirdy et al. 2010). Organic-rich and anoxic sediment conditions promote formation of Monosulfidic Black Oozes (MBOs) at or near the sediment surface which are now prevalent in the Coorong (Fitzpatrick et al. 2018). The hypersaline conditions and anoxic sediment are also likely to inhibit microbial processes that influence nutrient cycling. Specifically, the combination of organic rich, anoxic sediments and low nitrate (i.e. high ratio of electron donors to acceptors) would favour DNRA over coupled nitrification-denitrification as a nitrate reduction process, which we hypothesise is increasing the efficiency of N retention and NH_4^+ flux back to the water column (Kemp et al. 1990, Nizzoli et al 2000, Hardison et al 2015). The potential for cyanobacterial fixation of atmospheric N_2 to form bioavailable ammonia/ammonium (Paerl 2017) is also as yet unconstrained in the Coorong but may not be important given the limited dissolved P. There may also be potential for carbonate minerals (e.g. calcium carbonate, $CaCO_3$) to bind phosphorus following dissolved carbonate precipitation in hypersaline waters (Shao et al. 2018). Groundwater nutrient inputs are currently poorly known (Haese et al. 2008). Elucidating the above potential internal drivers of eutrophication requires specific process-based and integrated biogeochemical research, including focus on *in-situ* microbial processes. This is occurring as part of the U*nderstanding Nutrient Dynamics* component of the HCHB Trials and Investigations project.

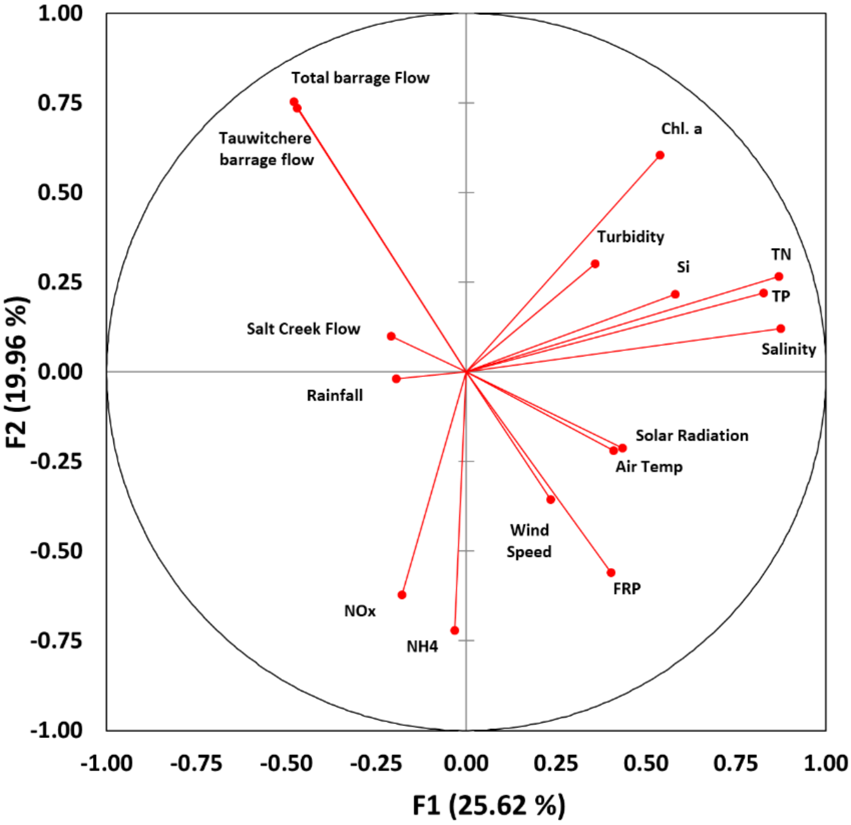


Figure 6. PCA loading plot for Factor 1 (F1) and Factor 2 (F2) comprising water quality (Salinity/EC, TN, TP, NH_4, NO_x, FRP, reactive Si, chlorophyll a, turbidity) , hydrological (Salt Creek inflow, total barrage inflows, Tauwitchere barrage inflows), and meteorological (wind speed, solar radiation, air temperature, rainfall) variables.

Table 1: Squared cosines of the Principal Component Analysis variables and factors (F1-F3). Values in bold correspond for each variable to the factor for which the squared cosine is the largest.

VARIABLE	F1	F2	F3
NH_4	**0.001**	0.519	**0.141**
Chlorophyll *a*	**0.290**	0.365	**0.084**
NOx	**0.032**	0.385	**0.083**
TN	0.755	**0.071**	**0.006**
FRP	**0.162**	0.313	**0.053**
TP	0.684	**0.049**	**0.040**
Reactive Si	0.338	**0.047**	**0.122**
Salinity	0.764	**0.015**	**0.006**
Turbidity	0.129	**0.091**	**0.011**
Salt Creek Flow	**0.044**	**0.010**	0.136
Total Barrage Flow	**0.229**	0.568	**0.001**
Tauwitchere Barrage Flow	**0.222**	0.541	**0.006**
Solar Radiation	**0.189**	**0.045**	0.417
Rainfall	**0.038**	**0.000**	0.423
Air Temp	**0.168**	**0.048**	0.569
Wind Speed	**0.055**	0.127	**0.004**

Another key question in regard to drivers of eutrophication of the Coorong is whether any particular nutrient is limiting for phytoplankton growth (i.e. autochotonous organic matter production/eutrophication). A plot of TN vs TP, and TN and TP versus Chlorophyll *a* reveals that ratios are quite variable but some general patterns emerge (Figure 7). Firstly, the TN to TP ratio is typically well above the Redfield Ratio, which suggests P limitation is present. However, this apparent P limitation could be driven by N being in excess. Secondly, a plot of TP vs chlorophyll *a* against the global estuary N and P limitation regression models of Smith (2006) also suggests phytoplankton P limitation is much more likely in the Coorong. Lastly, the likelihood of P limitation is consistent with the very low FRP relative to the dissolved nitrogen species (Figure 4). The potential over-supply of N driving this apparent P limitation requires further assessment as part of specific nitrogen process based research. Reactive Si does not appear to be limiting with concentrations in the range of ≈40 µM.

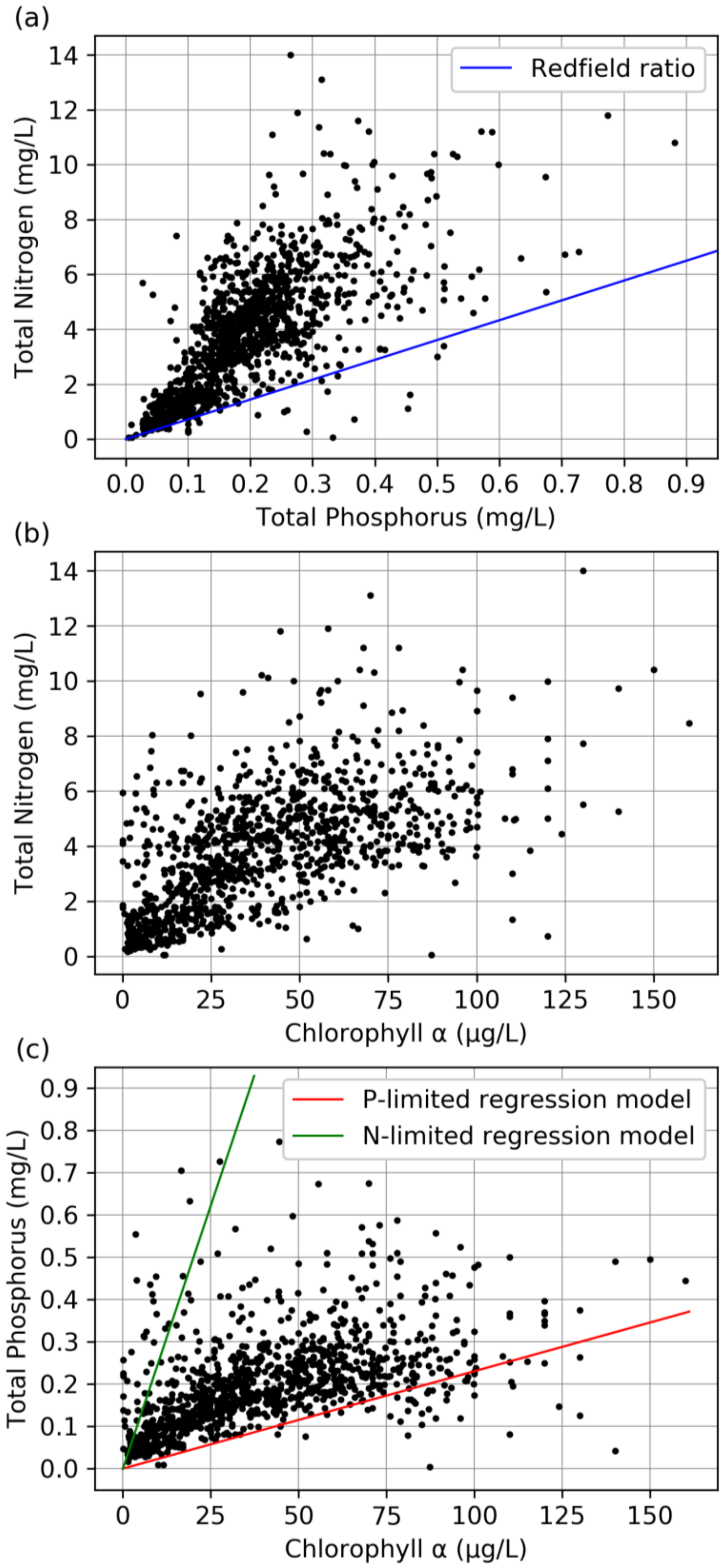


Figure 7. Relationships between total nitrogen, total phosphorus and Chlorophyll *a* in the Coorong. The Redfield Ratio (16N:1P in molar units or 7.2N:1P in these mass units) is shown on the TN vs TP plot with the nutrient limitation relationships from Smith et al. (2006) displayed on the TP vs Chlorophyll *a* plot.

3.4 Ecosystem impacts and feedback loops arising from hypersalinisation and eutrophication

Large ecosystem state changes have occurred in the Coorong since the 1950s, where slower growing *Ruppia* sp. seagrasses have been replaced by fast growing phytoplankton and filamentous algae (Krull et al. 2009, Dick et al. 2011, Kim et al. 2013, Brookes et al. 2018, Collier et al. 2019). While the main seagrass species now present, *Ruppia tuberosa*, is salt tolerant, its salinity tolerance has been regularly exceeded (e.g. EC > 105-108 mS/cm or salinity > 85-90 psu for seed germination; Kim et al. 2013). The seagrass *Ruppia megacarpa* has now been lost from the Coorong as this has a much lower salinity tolerance (Krull et al. 2009, Dick et al. 2011, Kim et al. 2013).

Along with extreme hypersalinity, it is also highly likely that eutrophication has contributed to the decline in *Ruppia* ecological condition. Nutrients are now being retained in the system and partitioned into high turnover organic phases. The persistently high organic-derived turbidity is likely limiting light penetration and hence *Ruppia* habitat suitability. Research from other eutrophic lagoons has also shown that eutrophication and high organic loadings to the sediment can lead to anoxic conditions and toxic sulfide build up resulting in loss of *Ruppia* (Heijs et al. 2000, Azzoni et al. 2001).

Increasing hypersalinisation in the South Lagoon and southern region of the North Lagoon has also resulted in the complete loss of benthic macroinvertebrate communities (Dittmann et al. 2015, Tweedley et al. 2019). This is due to the salinity tolerances (EC > 77 mS/cm or salinity ≈ 60 psu; equation 1) of key species being persistently exceeded (Remailli et al. 2018, Dittmann et al. 2015). Loss of these benthic macroinvertebrates and their ecological functions may have had profound influences on internal nutrient cycling and water quality. Burrowing and bioturbating invertebrates oxygenate the sediment, enabling the formation of iron oxides (which sequester phosphate), and stimulating rates of coupled nitrification-denitrification reactions that promote nitrogen loss as gaseous end-products (Welsh 2000, Stief 2013). Thus, loss of benthic infauna due to hypersalinisation is likely to have contributed to the increasing eutrophication by favouring nutrient recycling over nutrient elimination/sequestration processes. Capitellid polychaetes, which are an indicator for polluted and eutrophic sediment conditions (Cardoso et al. 2007) are dominating the benthic community in the North Lagoon (Dittmann et al. 2018).

3.5 Application of findings to management of the Coorong

This section presents the application of the findings of this work to the management of the Coorong at a conceptual level based on the current understanding of the system. There are significant uncertainties related to the relative importance of internal and external nutrient sources that currently limit our capacity to predict the water quality responses to specific management interventions. Indeed, addressing these uncertainties is a key purpose of the *Understanding Nutrient Dynamics* component of the Trials and Investigations project of HCHB.

Overall, reduced flushing of the Coorong lagoon system appears to be a key determinant of the degree of eutrophication, which is generally consistent with global evidence from other estuaries (Bricker et al. 1999, Nixon 2009, Le Moal et al. 2019). Higher salinity conditions, increasing above seawater values, were associated with a higher degree of eutrophication and increasing TN, TP and chlorophyll *a* concentrations (Figure 5). The Coorong, with its reverse estuary character, is undoubtedly a system with a high propensity to retain nutrients and appears highly sensitive to reductions in flushing. This has likely been exacerbated by periodic drought conditions followed by a decade long hydrological drought due to increasing water extraction and diversion in the Murray-Darling Basin over the last century (Mosley et al. 2012). In the absence of sufficient freshwater inflows and adequate flushing, the effects of evaporative concentration of salts and other constituents appears quite extreme in the South Lagoon due to its closed morphology. This could be exacerbated by predicted future temperature increases, rainfall decreases, and flow decreases due to climate change in the Murray-Darling Basin (CSIRO 2018).

Increasing system flushing (frequency and magnitude), in particular for the South Lagoon, to try and reverse eutrophication would, in theory, be beneficial to: (a) export nutrients, algae and organic matter; (b) reduce algae and total nutrient concentrations in the water column to reduce deposition of organic matter and nutrients to the sediment; (c) reduce algal-derived turbidity to enable increased light penetration for seagrasses; (d) reduce hypersalinisation to enable re-establishment of benthic macroinvertebrates, and; (e) reduce formation of hypersaline, reduced, sulfide-rich sediments that we hypothesise are inhibiting healthy nutrient cycling.

Two options available to increase flushing of the Coorong are: (1) increase seawater inflows; and/or (2) increase freshwater inflows. Conceptual models for these options are shown in Figure 8 alongside a conceptual model of the current persistent hypersaline and hypereutrophic state. At a conceptual level, increased seawater flushing offers potential benefits as nutrient concentrations and turbidity are lower in the coastal seawater. Increased seawater flushing could also likely achieve and sustain a higher level of dilution and nutrient export as it is not constrained by climatic factors (e.g. droughts) and Murray-Darling Basin water allocation decisions. Achieving increased seawater flushing of the South Lagoon would require: (a) new infrastructure for siphoning, channelling or pumping of seawater directly into the South Lagoon; and/or (b) significant dredging works to remove sedimentation in several locations from the Murray Mouth (Bourman et al. 2018) and between the two lagoons. These interventions are being considered by the Healthy Coorong Healthy Basin as part of the Coorong Infrastructure Investigations Project (CIIP) which has the objective of investigating the feasibility of multiple long-term operational infrastructure and management options to improve the ecological health of the Coorong. If this type of intervention was undertaken, the desired salinity regimes and environmental gradients would require careful assessment and controlled operations to take into account the natural salinity gradients, while maintaining and restoring key ecosystem functions in this Ramsar-listed ecosystem.

The importance of freshwater inflows from the River Murray to maintaining suitable water quality in the Coorong has been long known (Geddes and Butler 1984) and the success of any future management interventions will also be dependent upon adequate inflows. The increased freshwater flushing option (Figure 8) would not necessarily require new infrastructure; however, securing regular delivery of large volumes of environmental water from the River Murray above that of the current Murray-Darling Basin Plan to achieve this may be difficult. The early study of Geddes and Butler (1984) demonstrates large River Murray flood events can markedly reduce salinity and eutrophication throughout the Coorong. There is potential additional water available from the South-East catchment that could be redirected to Salt Creek during winter-spring periods. However as both the River Murray-Lower Lakes and Salt Creek freshwater inputs have higher nutrients and turbidity than seawater, the effects of increased external nutrient loading needs to be balanced against the flushing effects of the additional flows. It should be noted the increased seawater and freshwater flushing options are not mutually exclusive. Indeed, a combination of both increased freshwater and seawater inflows over the longer term may be the preferred management approach, as this is likely closer to the natural state of the Coorong.

Complementary actions could include aquatic plant and benthic invertebrate restoration, and localised sulfidic sediment remediation (Sullivan et al. 2018), but these may be challenging to implement at the lagoon system scale and unlikely to be successful over the longer term unless overall system-wide water quality can be improved and maintained. There are unlikely to be “quick fixes” to the current degraded state of the Coorong. Reducing eutrophication could take years to decades as there are internal loadings of nutrients that will take time to reduce, even if the nutrients from catchment sources and water column eutrophication are reduced.

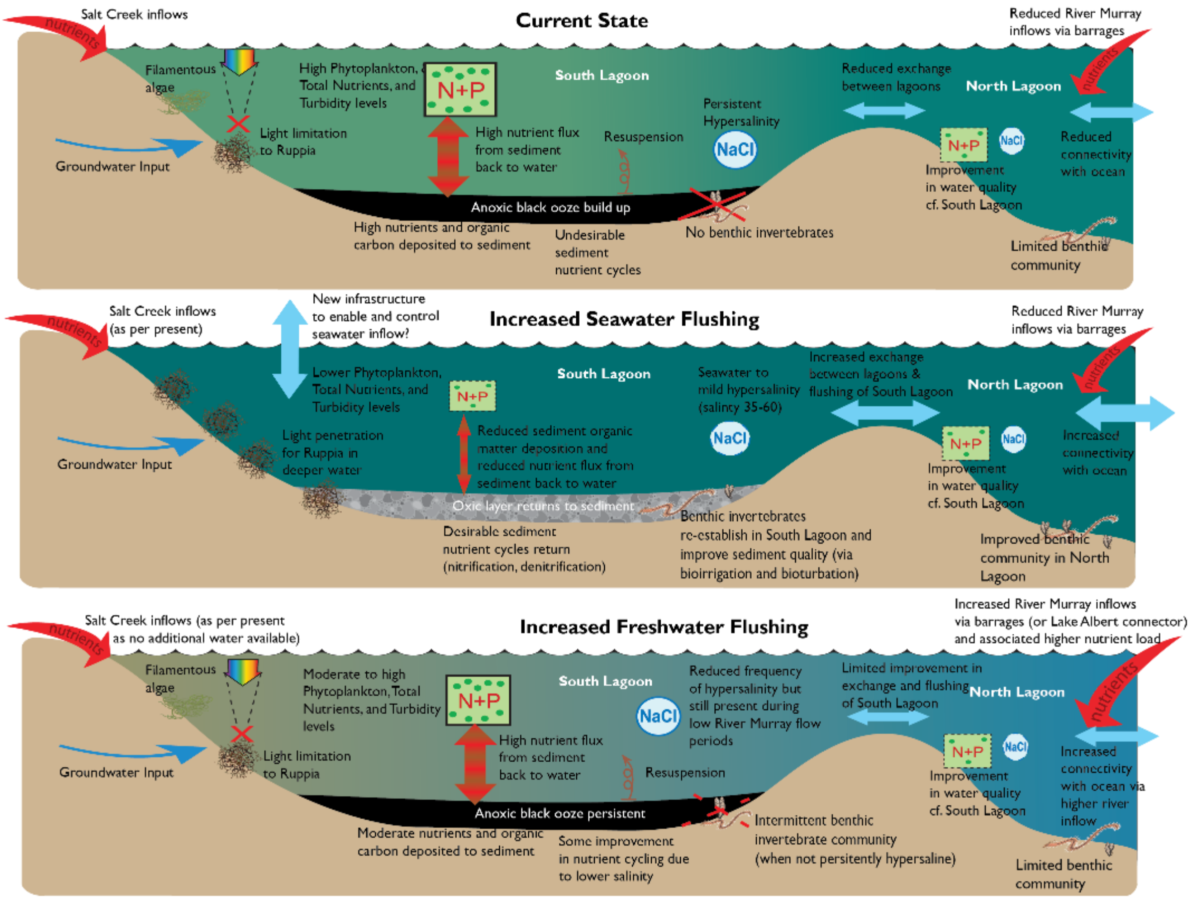


Figure 8. Conceptual model of the current water quality state in the Coorong (top) and with increased seawater (middle) and increased freshwater flushing (bottom).

4 Conclusions

The Coorong has experienced major issues over several decades with more frequent hydrological drought leading to reduced flushing, hypersalinity, hypereutrophication, and loss of the keystone species. Many other estuarine systems globally are facing increased pressure and poorer water quality outcomes due to the effects of water extraction and climate change. The Coorong provides a good example of an extreme manifestation of pressures that are occurring globally. Much of the Coorong (especially the South Lagoon) is now in a persistent hypereutrophic and hypersaline state. Evidence suggests that changed hydrodynamics and evapo-concentration processes in the Coorong has reduced flushing and driven high concentrations of salinity, total nutrients and phytoplankton. Although eutrophication in the Coorong is overall enhanced by hypersalinisation, the effects are non-linear. Hypersalinisation likely reinforces eutrophication process by negatively impacting benthic fauna and flora communities, which promote nutrient sequestration and elimination processes. Research in the *Understanding Nutrient Dynamics* component of the HCHB Trials and Investigations will help to better understand the internal nutrient cycling and causal linkages between salinity and nutrients.

The Coorong, by the nature of its location at the end of a large river catchment in arid to semi-arid settings with major water management challenges, is highly susceptible to reduced flushing, evaporation and resultant impacts. Achieving better flushing of the system via increased seawater and/or freshwater inflows, and lagoon hydraulic connectivity, would be desirable to lower and manage hypersalinity and eutrophication. The degree of dilution and nutrient export that could be achieved due to increased freshwater flushing is limited by volumes of environmental water available in the Murray-Darling Basin and comes with cost of increased external nutrient loading, whereas increased seawater flushing, via more seawater inflow and/or greater connectivity, could in theory provide a more reliable means of nutrient-organic matter dilution and export. Process-based research to improve understanding of internal nutrient dynamics (e.g. sediment-water nutrient fluxes, N fixation, coupled nitrification-denitrification and DNRA processes, and influence of bioturbation and irrigation by macroinvertebrates) is urgently required to better understand the drivers of eutrophication in this system. This being undertaken in the *Understanding Nutrient Dynamics* component of the HCHB Trials and Investigations project, and is an essential step before any inflow management option is considered. Restoration of water quality of this unique and important ecosystem is likely to take some time due to internal nutrient loadings and current loss of key habitats and associated environmental processes.

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Appendix

Table A1.1: List of water quality monitoring sites in the Coorong. The coordinates are in the Geocentric Datum of Australia 1994 (GDA 94), Zone 54.

NAME	EASTING	NORTHING
Murray Mouth	308015	6063155
Tauwitchere	320218	6059692
Mark Pt	325762	6054914
Long Point	333756	6048260
Noonameena	342849	6041481
Bonneys	347969	6037304
McGrath Flat Nth	354600	6029390
Parnka Pt	355237	6025730
Villa de Yumpa	359175	6022894
Stony Well	365104	6017790
North Jacks Pt	369342	6010972
South Policemans Pt	372453	6005680
Snipe Point	374406	6002900
1.8km west of Salt Creek	375882	6000470
South Salt Creek	377463	6000594
3.2km south of Salt Ck	377570	5997290

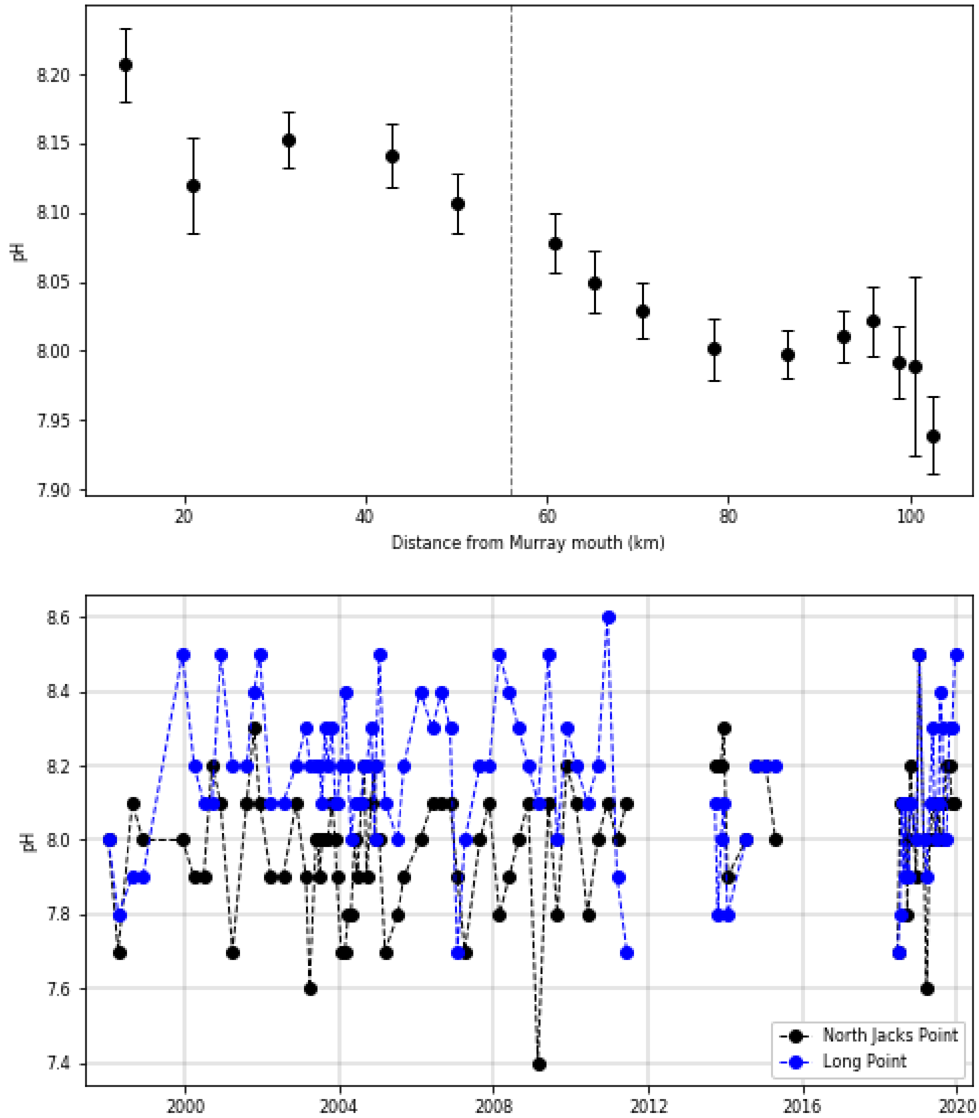

Figure A1.1. Changes in pH plots along the Coorong and across time.

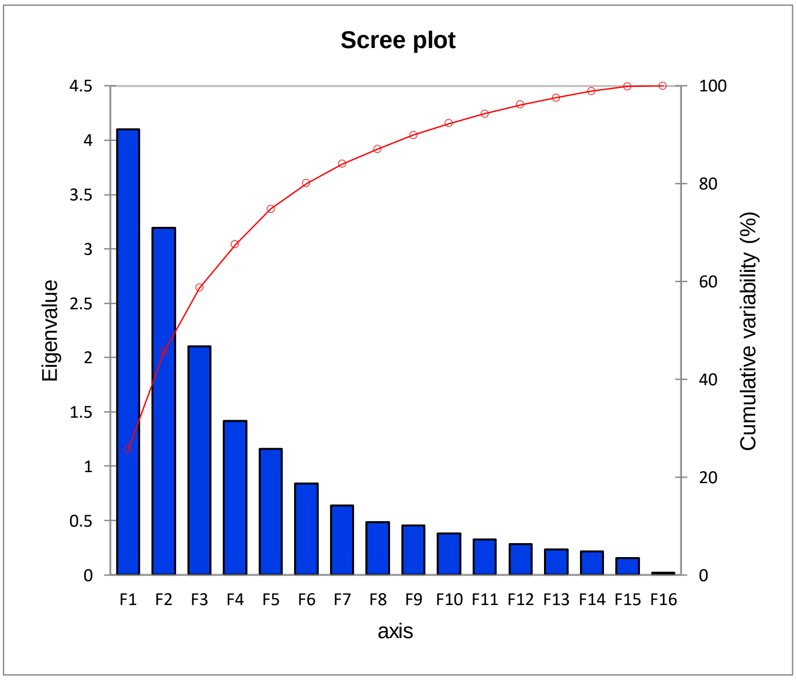

Figure A1.2. Principal Component Analysis Scree Plot.

Table A1.2: Correlation (Spearman R) matrix from the Principal Component Analysis. Bold values are those significant at a 0.05 level.

VARIABLES	NH4	CHL. A	NOX	TN	FRP	TP	SI	SALINITY	TURBIDITY	SALT CREEK FLOW	BARRAGE FLOW	TAUWITCHERE FLOW	SOLAR RADIATION	RAINFALL	AIR TEMP	WIND SPEED
NH4	1	-0.305	**0.641**	**-0.157**	**0.493**	**-0.091**	0.007	**-0.093**	**-0.123**	**0.092**	**-0.399**	**-0.408**	**-0.078**	**0.112**	-0.056	**0.134**
Chl. a	-0.305	1	**-0.328**	**0.600**	-0.022	**0.570**	**0.514**	**0.490**	**0.397**	-0.045	**0.186**	**0.149**	**-0.093**	**0.077**	-0.058	**-0.101**
NOx	0.641	-0.328	**1**	**-0.278**	**0.363**	**-0.163**	**-0.132**	**-0.218**	-0.017	0.072	**-0.219**	**-0.211**	-0.050	**0.105**	-0.065	**0.105**
TN	-0.157	0.600	**-0.278**	**1**	**0.206**	**0.750**	**0.543**	**0.800**	**0.373**	-0.074	**-0.198**	**-0.191**	**0.275**	**-0.134**	**0.228**	**0.111**
FRP	0.493	**-0.022**	**0.363**	**0.206**	**1**	**0.229**	**0.231**	**0.237**	0.011	0.075	**-0.463**	**-0.485**	**0.226**	**-0.082**	**0.109**	**0.148**
TP	-0.091	0.570	**-0.163**	**0.750**	**0.229**	**1**	**0.487**	**0.736**	**0.457**	-0.072	**-0.204**	**-0.204**	**0.177**	-0.066	**0.131**	**0.156**
Si	**0.007**	0.514	**-0.132**	**0.543**	**0.231**	**0.487**	**1**	**0.540**	0.017	**0.090**	**-0.091**	**-0.105**	0.005	-0.038	-0.029	-0.070
Salinity	-0.093	0.490	**-0.218**	**0.800**	**0.237**	**0.736**	**0.540**	**1**	**0.224**	**-0.214**	**-0.332**	**-0.315**	**0.251**	-0.072	**0.253**	**0.138**
Turbidity	-0.123	0.397	-0.017	**0.373**	0.011	**0.457**	0.017	**0.224**	**1**	**0.077**	**0.109**	**0.097**	**0.139**	-0.031	0.013	**0.090**
Salt Creek Flow	0.092	**-0.045**	0.072	-0.074	0.075	-0.072	**0.090**	**-0.214**	**0.077**	**1**	**0.194**	**0.125**	**-0.108**	0.029	**-0.379**	**-0.212**
Barrage Flow	-0.399	0.186	**-0.219**	**-0.198**	**-0.463**	**-0.204**	**-0.091**	**-0.332**	**0.109**	**0.194**	**1**	**0.972**	**-0.254**	-0.020	**-0.271**	**-0.382**
Tauwitchere Flow	-0.408	0.149	**-0.211**	**-0.191**	**-0.485**	**-0.204**	**-0.105**	**-0.315**	**0.097**	**0.125**	**0.972**	**1**	**-0.256**	-0.039	**-0.231**	**-0.291**
Solar Radiation	-0.078	-0.093	-0.050	**0.275**	**0.226**	**0.177**	0.005	**0.251**	**0.139**	**-0.108**	**-0.254**	**-0.256**	**1**	**-0.436**	**0.657**	**0.082**
Rainfall	0.112	0.077	**0.105**	**-0.134**	**-0.082**	-0.066	-0.038	-0.072	-0.031	0.029	-0.020	-0.039	**-0.436**	**1**	**-0.500**	**0.205**
Air Temp	**-0.056**	**-0.058**	-0.065	**0.228**	**0.109**	**0.131**	-0.029	**0.253**	0.013	**-0.379**	**-0.271**	**-0.231**	**0.657**	**-0.500**	**1**	**0.115**
Wind Speed	0.134	-0.101	**0.105**	**0.111**	**0.148**	**0.156**	-0.070	**0.138**	**0.090**	**-0.212**	**-0.382**	**-0.291**	**0.082**	**0.205**	**0.115**	**1**

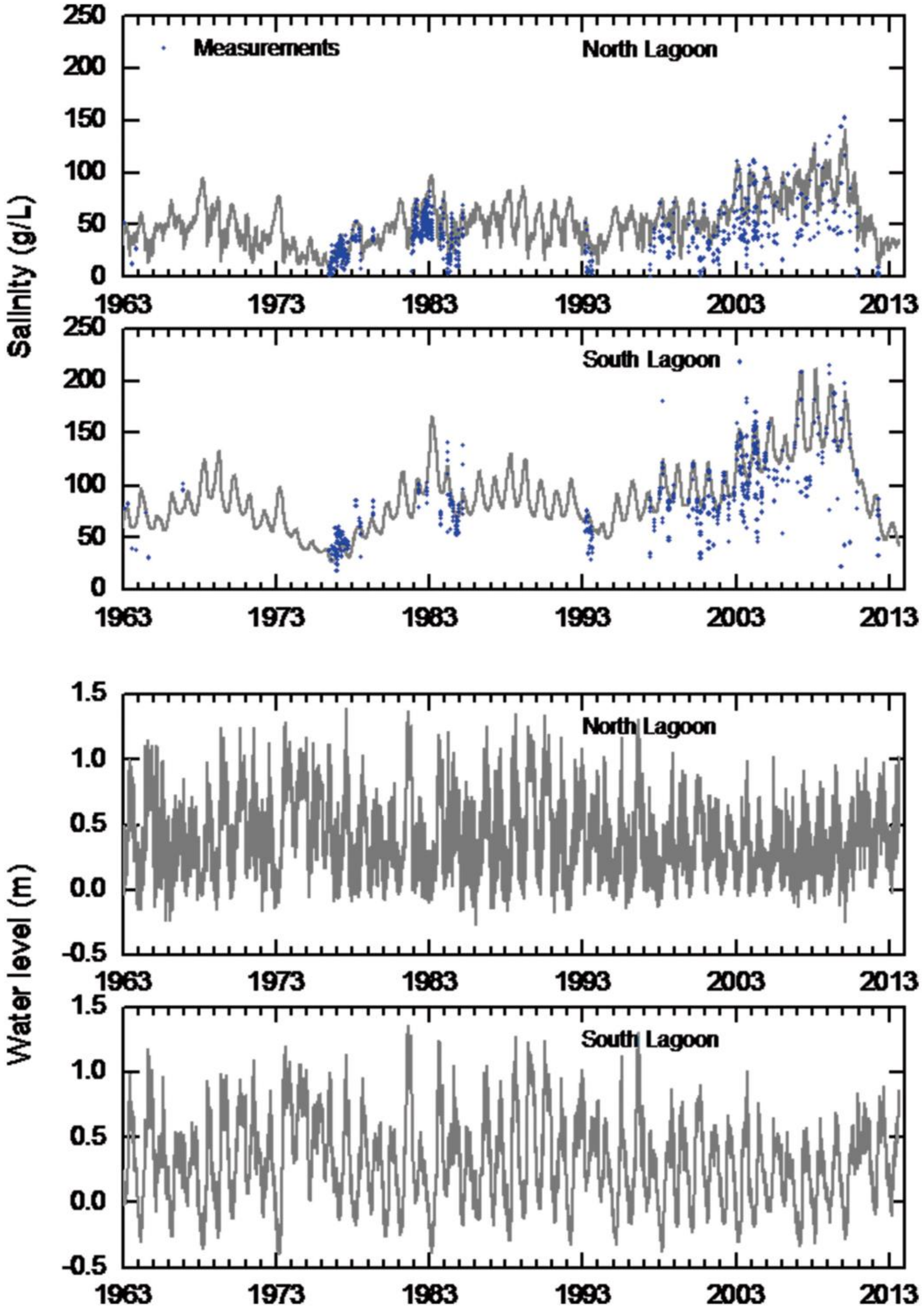

Figure A1.3. Measured and modelled salinity of the Coorong from 1960-2018 (from Gibbs et al. 2018).

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