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Long-term study of phytoplankton dynamics in a supply reservoir reveals signs of trophic state shift linked to changes in hydrodynamics associated with flow management and extreme events

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ABSTRACT

This study analyses over a decade (2009–2022) of monitoring data to understand the impact of hydrological characteristics on water quality and phytoplankton dynamics in Prospect Reservoir, a critical water supply for Greater Sydney, Australia, known for its excellent water quality. Water quality and phytoplankton dynamics were related to hydrodynamics, linked to flow management and the water quality of inflows. Phytoplankton biovolume increased after a prolonged drawdown and subsequent refill event, mainly driven by dinoflagellates, and corresponded to increases in total phosphorus and water temperature. The hydrological period following the 2019/2020 summer bushfires (post-bushfire) that impacted connected reservoirs, was marked by increased flow activity and nutrient loading, leading to significant shifts in the phytoplankton community. Functional group classification and ordination analysis indicated a transition from taxa typically dominant in oligotrophic conditions to meso-eutrophic. This transition correlated with elevated nutrient levels and chlorophyll-a (Chl-a), and reduced Secchi depth and dissolved oxygen, providing evidence of eutrophication. Q index indicated good water quality post-bushfire, contrasting with a eutrophic status assessment using Chl-a. Our findings highlight the importance of analysing long-term datasets encompassing varied hydroclimatological conditions for a deeper understanding of reservoir behaviour. A comprehensive approach to water quality assessment is recommended, combining functional group classification, Q index and Chl-a measurements for effective reservoir health assessment. This research provides novel insights into the effects of disturbances such as bushfires, on water quality and phytoplankton dynamics in an underrepresented geographic region, offering valuable knowledge for managing water resources amidst growing climate variability.

1. Introduction

Phytoplankton are essential to aquatic ecosystems, serving as a base energy source. However, their overabundance can pose issues for water treatment and some taxa can compromise drinking water quality by producing toxins or taste and odour compounds (Merel et al., 2013). Reservoir operations and the hydrodynamic features of a waterbody can strongly influence the conditions in the water column such as light, nutrient concentrations and water temperature. These are important factors which can affect phytoplankton growth and community composition (Reynolds et al., 2002) and under certain conditions this may lead to problematic groups such as cyanobacteria dominating. The degradation of water supplies by nuisance taxa can result from changes in hydrodynamics. These changes can be linked to inflow/outflow (flow) regulation, catchment land-use, and climate change, which are projected to compound and intensify in the future (Foley et al., 2005; Paerl and Huisman, 2009). Addressing these challenges and ensuring effective water management requires understanding the factors contributing to the proliferation of nuisance taxa. Strategies aimed at reducing water residence time by increasing flow activity can be useful for managing phytoplankton growth by flushing excess nutrients and nuisance taxa (Fornarelli and Antenucci, 2011; Lucas et al., 2009; Mitrovic et al., 2001; Padovesi-Fonseca et al., 2009; Watts, 2000). However, the success of the strategies is contingent upon various interrelated factors including the quantity, quality and frequency of flows, water level changes, and the morphological and ecological characteristics of the waterbody,

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underscoring the complexity in managing aquatic systems.

One key target area for water managers is controlling the supply of nutrients for phytoplankton growth. Phosphorus and nitrogen (macronutrients) are often the most important limiting nutrients driving phytoplankton growth in freshwater systems (Mueller and Mitrovic, 2015; Paerl et al., 2016). Additionally, trace metals such as iron, manganese, zinc and cobalt, are essential for biological processes such as photosynthesis and protein synthesis (Facey et al., 2019). A recent study by Facey et al. (Facey et al., 2021) has suggested they may influence phytoplankton dynamics in some south-eastern Australian waterbodies. Nutrient enrichment (eutrophication) is linked to the dominance of cyanobacteria, producers of taste and odour compounds and a range of toxins that can have adverse effects on public health and local biodiversity (Merel et al., 2013). Furthermore, eutrophication typically increases phytoplankton abundance, impacting water treatment by raising filtering and coagulation costs (Joh et al., 2011). Consequently, managing waterbodies to maintain nutrient-deficient (oligotrophic) conditions is preferable for water use (ANZECC and ARMCANZ 2000).

Extreme events, such as droughts conducive to uncontrolled bushfires, may elevate both macronutrient and trace metal input into aquatic systems (Paul et al., 2022; Sánchez-García et al., 2023). These disturbances further complicate water management efforts, especially when they are followed by heavy rainfall. Variability in weather patterns, such as the El Niño Southern Oscillation phenomenon which affects regions including south-eastern Australia, contributes to these weather extremes (Min et al., 2013). Despite the known effects of extreme weather events, particularly bushfires on nutrient dynamics (Paul et al., 2022; Sánchez-García et al., 2023), research on their impact on phytoplankton dynamics remains limited. Most studies have focused on North American waterbodies and river systems, with varying findings and a lack of assessments on community changes (Paul et al., 2022). As climate change is expected to increase the frequency and severity of such extreme events (Fischer and Knutti, 2015), comprehensively understanding their impacts to water resources is imperative (Khan et al., 2015).

Nutrient concentrations are commonly used indicators of water quality; however, biotic indices may be more sensitive in detecting changes (Carvalho et al., 2013; Strobl and Robillard, 2008). Due to their short generation times, phytoplankton can act as short-term biological indicators, signaling shifts in water quality, and potentially providing early evidence of trophic state changes and water pollution (Reynolds, 1998). Chlorophyll-*a* (Chl-*a*) is commonly used as a surrogate for lake productivity and is typically a reliable indicator of trophic status (Lyche-Solheim et al., 2013). However, being non-specific and influenced by algal composition and cell physiology (Bowles, 1982), it cannot differentiate phytoplankton groups or identify nuisance species. Phytoplankton taxa can be grouped into assemblages, or functional groups, based on adaptations to specific environmental conditions (Padisák et al., 2009; Reynolds et al., 2002). The Q index, derived from this grouping system (Padisák et al., 2006), has been effectively used to assess water quality of waterbodies of varying hydroclimatological characteristics (Becker et al., 2009; Becker et al., 2010; Cellamare et al., 2012; Crossetti and Bicudo, 2008).

Prospect Reservoir is an earth embankment dam located at the primary water filtration plant that supplies potable water for over 5.3 million residents in the Sydney metropolitan region of Australia. In the past decade modifications to Prospect's supply operations have resulted in dynamic changes to its hydrology. This included an extended drawdown period for dam maintenance and changes in inflow regularity and quality from connected reservoirs. Given the importance of Prospect Reservoir for Greater Sydney's drinking water supply, this study seeks to analyse over a decade of monitoring data to 1) examine the relationship between hydrology, water quality and phytoplankton dynamics, 2) evaluate the use of functional group classification to explain environmental changes, and 3) assess trophic state indicators for lake management.

2. Materials and method

2.1. Study site

Prospect Reservoir is the largest freshwater reservoir in the Sydney region (latitude −33.821; longitude: 150.894), with a lake surface area of approximately 5.8 km^2, and maximum and mean depth of 20 m and 9 m respectively. Inflows to Prospect Reservoir come from Warragamba Dam via the Warragamba pipeline, and the Upper Nepean Dams (Cataract, Cordeaux, Avon, and Nepean) via the Upper Canal. Historically, drinking water was drawn from Prospect Reservoir, which was dosed directly with alum as part of the treatment process. A water filtration plant and bypass channel were commissioned in 1996, and since then water can be sent directly to the filtration plant from the Warragamba pipeline and Upper Canal (Fig. 1), resulting in no inflows to Prospect Reservoir for periods of time. When water quality from the main supply sources is impacted, inflows can be redirected into Prospect Reservoir for dilution or supply can be drawn directly from Prospect Reservoir, which occurred almost constantly from 2020 to 2023. Prospect Reservoir therefore remains key to the resilience of Greater Sydney's water supply system, for example during flood events, seasonal lake turnover and infrastructure maintenance activities.

Prospect Reservoir has a clear and well-mixed water column, an extensive bed of submerged macrophytes, and was historically characterised as oligotrophic based on total phosphorus (TP) readings from 2000 to 2010 and mesotrophic using Chl-*a* readings from the same period. This well-mixed condition is attributed to an aerator, which remained continuously operational for most of the study period. The catchment is relatively small (9.7 km^2), with the lake surface area constituting more than half (59.8 %). The rest of the catchment consists almost entirely of native vegetation. Thus, due to its size and composition, catchment inflow is not expected to be an important hydrological factor influencing Prospect Reservoir's water quality. The submerged macrophyte community is predominantly native species and include *Chara* spp., *Hydrilla verticillate, Najas tenuifolia, Potamogeton* spp., and *Vallisneria gigantea.*

2.2. Dataset and sampling methods

A long-term hydrometric, water quality and phytoplankton monitoring dataset (2009–2022) was obtained from WaterNSW. Prospect Reservoir storage level was measured at 15-min intervals with a telemetered pressure transducer and converted to mean daily storage level in meters below full storage level. Telemetered flow meters were used to measure Warragamba Pipeline and Upper Canal discharges at 5-minute intervals and converted to mean daily discharge for each inflow. Daily precipitation data (mm) were obtained from the Prospect Reservoir weather station provided by the Bureau of Meteorology to calculate annual rainfall. For missing data, the next closest weather station (Seven Hills) was used.

Secchi depth (SD) measurements, surface water and Chl-*a* composite samples (0–6 m) were collected monthly at the mid-lake site (RPR1, Fig. 1), although some months are missing during early years of the time series. In total 154 samples were collected over the monitoring period. Physical and chemical field variables and phytoplankton composite samples (0 – 6 m) were generally sampled monthly during cooler seasons (June to September) and biweekly to weekly during warmer seasons (October to May), providing 903 phytoplankton community, and 791 physical and chemical, sample sets. Values that were measured below the limit of detection (LOD) were replaced with half the LOD concentration (LOD/2).

SD was measured using a Secchi disc. Water chemical samples were collected and preserved according to standard methods, including filtering with a 0.45 μm pore size glass fibre filter for dissolved nutrients (APHA 2017). Chl-*a* samples were filtered with a 1.2 μm pore size glass fibre filter within 24 h of collection and extracted in 94.5 % cold

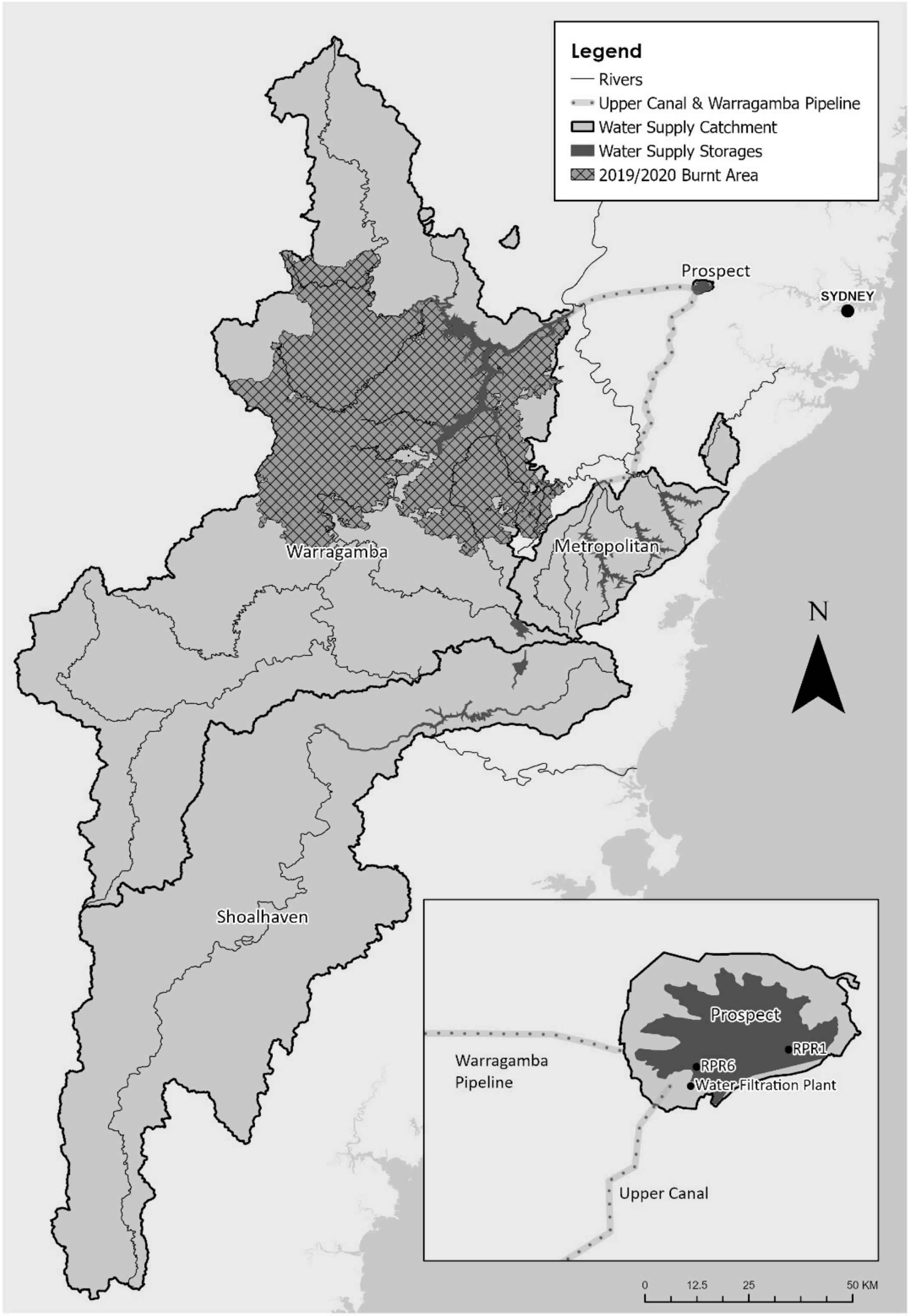

Fig. 1. Key water storages for the Sydney region, New South Wales, featuring the catchments of Prospect Reservoir, Warragamba (Warragamba Dam), Shoalhaven and Metropolitan (Upper Nepean Dams). The extent of warragamba catchment burnt during the 2019/2020 Summer Bushfires is marked. An inset of Prospect Reservoir's catchment details the location of the mid-lake sampling site (RPR1), inflow sources (Warragamba Pipeline and Upper Canal), and the water filtration plant: located directly next to the outlet.

acetone. Chl-*a* extracts and water chemical grab samples were stored at −18 °C until analysis. Samples were analysed at National Association of Testing Authorities (NATA) accredited facilities by standard methods (APHA 2017); Chl-*a* by method 10 200, dissolved organic carbon (DOC) by method 5310 B, total nitrogen (TN) and total phosphorus (TP) by methods 4500-N B and 4500-P I, Silica (Si) by method 4500-SiO2 D, and total iron (Fe), manganese (Mn) and aluminium (Al) by method 3125. Alkalinity and suspended solids were analysed using methods referenced to 2540 D and 2320 B respectively (APHA 2017).

Physical and chemical field measurements collected include water temperature (WT), dissolved oxygen (mg/L) and dissolved oxygen (% sat) (DO), pH and electrical conductivity (EC). Phytoplankton samples were preserved with Lugol's solution immediately after collection and were enumerated under an upright light microscope using a Lund cell (Hötzel and Croome, 1999). Taxa were identified to at least genus level, and to species level where possible. Algal cells were equated to a geometric shape to determine their surface area (representing the maximal cross-sectional area of the cell) and algal dimensions were measured to obtain a mean biovolume conversion factor for each taxonomic group. These factors were multiplied by the enumerated cell count to obtain biovolumes for each taxonomic group.

2.3. Data analysis

Trophic status was classified using mean annual Chl-*a* concentrations following Australian and New Zealand Environment and Conservation Council (ANZECC, 2000) guidelines (2000): < 2 μg/L: oligotrophic; 2–5 μg/L: mesotrophic; >5 μg/L: eutrophic. The assemblage index (Q Index), developed by Padisák et al. (Padisák et al., 2006), calculates the relative share (p_i, where $p_i = n_i/N$; n_i: biomass of the i th functional group; N: total biomass) of functional groups in total biomass and a factor number (F) determined for the i th functional group in the given lake type.

Factor F was assigned to each functional group based on the pristine status of the natural ecosystem and potential algal assemblages. Higher F values indicate groups found in pristine conditions, while lower values represent undesirable assemblages. Prospect Reservoir was assigned factor F specific to lake type 1 following Padisák et al. (Padisák et al., 2006). The Q index value can range from 0 – 5 and from this, water quality can be classified into five categories: 0–1: very poor; 1–2: tolerable; 2–3: medium; 3–4: good; 4–5: excellent.

The total daily nutrient load was the sum of the nutrient loading from the two main inflow sources. This was calculated by first estimating the loading of TP, TN, Mn, Fe and Al using an estimated daily nutrient concentration from inflow monitoring data (biweekly for fortnightly data for Warragamba pipeline and Upper Canal respectively) supplied by WaterNSW and multiplying the value by the daily inflow volume. A principal component analysis (PCA) was then performed with the five nutrient loads which showed the first axis explained 92.28 % of variance. Thus, PC1 values were used to represent the total daily nutrient load which incorporated all five nutrients (TP, TN, Mn, Fe and Al). Water residence time was calculated using the formula: (mean annual storage volume)/(annual inflow or outflow volume), following Rueda et al. (Rueda et al., 2006). Given the significant variation between inflow and outflow volumes in this reservoir, with instances of zero annual outflow, we included both inflow and outflow in the calculation. Like our approach for nutrient load estimation, a combined inflow/outflow value was calculated using PCA. The first of the PCA explained 80.32 % of variance.

Time series trends of environmental variables, taxonomic phytoplankton biovolume and trophic indicators were detected using generalised additive mixed models (GAMM) and statistically significant change were analysed using the mgcv package within RStudio (version 4.3.1). Periods of significant change were identified as those time points for which the confidence interval of the first derivative did not include zero, with the first derivative calculated using the method of finite differences (Simpson, 2018).

PCA was performed on the functional group biovolume data (n = 907) over the whole study period. Detrended correspondence analysis (DCA) was performed on functional group biovolume data to determine whether linear or unimodal ordination methods should be used in the direct gradient analysis. The results from the DCA indicated that gradient lengths did not exceed 2.5 standard deviations units for all axes, suggesting a linear ordination was appropriate and thus, redundancy analysis (RDA) was utilised. To ensure multicollinearity did not affect the ordination, environmental variables with a variance inflation factor (VIF) >5 were removed. From this, dissolved oxygen (mg/L) (correlated with WT) and total aluminium (Al) (correlated with Fe) were removed from the analysis. Forward selection was performed to assess the significance of environmental variables on functional groups using Monte Carlo permutation (n = 999). Non-significant environmental variables pH, alkalinity (Alk) and suspended solids (SS) were removed from the final RDA plot. Functional group biovolume data and environmental variables were $\mathrm{Log}_{10}(x + 1)$ transformed prior to data analysis (with exception of pH). DCA and calculation of VIF were carried out in RStudio using the vegan and car packages. PCA, forward selection and RDA were run using CANOCO 5.15 package.

3. Result

3.1. Hydrology

Over the past 13 years, Prospect Reservoir had experienced changes to its hydrology that are closely linked to supply operations (Fig. 2). The study period was separated into six phases (Table 1).

3.2. Water quality changes

Analysis of long-term trends in environmental variables indicated significant fluctuations in some variables coinciding with hydrological/operational changes (Figs. 3 and 4). GAMM analysis revealed no significant changes in WT, TP and SS over the study period. The measured TP concentrations were typically 〈LOD of 0.005 mg/L. Measurements of TP 〉 LOD was however observed more frequently post-drawdown (Fig. 3). GAMM indicated a significant decreasing trend in Si linked with the post-drawdown period and a significant increasing trend was observed corresponding with an increase in flow activity. The mean annual Si measured in 2010 (0.875 ± 0.062 mg/L) was almost half the concentration measured in 2021 (1.56 ± 0.246 mg/L). Over the whole study period, Fe and SD were shown to have a significant increasing and decreasing trend, respectively. In 2013, during drawdown, the mean annual Fe (0.067 ± 0.007 mg/L) more than doubled compared to pre-drawdown levels (2010) (0.031 ± 0.004 mg/L). Post-bushfire (2021), the mean annual Fe (0.186 ± 0.03 mg/L) almost tripled in concentration compared to 2013. Yearly average SD was 2.89 ± 0.37 m in 2010, compared to 1.45 ± 0.16 m in 2021. Dissolved oxygen (%sat) (DO) and pH showed two periods of significantly decreasing trends, with the first ending post-drawdown, and the second corresponding with the post-bushfire (PBF) period. Nutrient variables including Mn and TN had a significant increasing trend which also corresponded with the PBF period.

3.3. Phytoplankton dynamics

There were two notable periods of increased phytoplankton biomass, driven by different phytoplankton phyla. The first was the drawdown event in 2012 which corresponded to a significant increase in dinoflagellate, diatom, cryptomonad and chrysophyte biovolume (Fig. 5). Elevated dinoflagellate biovolumes persisted until a year after the refill event, after which a decreasing trend began in 2017 and continued until 2021. Chrysophyte biovolume also began to significantly decline from 2018 until the end of the study period. The second period of increased phytoplankton biomass corresponded with the post-bushfire (PBF) period in 2019. Green algae were the primary drivers of this increase, and their growth corresponded with an increase in flow activity. Cryptomonad and diatom biovolume also contributed to this significant biovolume increase. GAMM indicated no significant increase in cyanobacteria biovolume over the whole study period, however, a significant seasonal trend was evident with higher biovolumes occurring during the warmer months (Fig. 5).

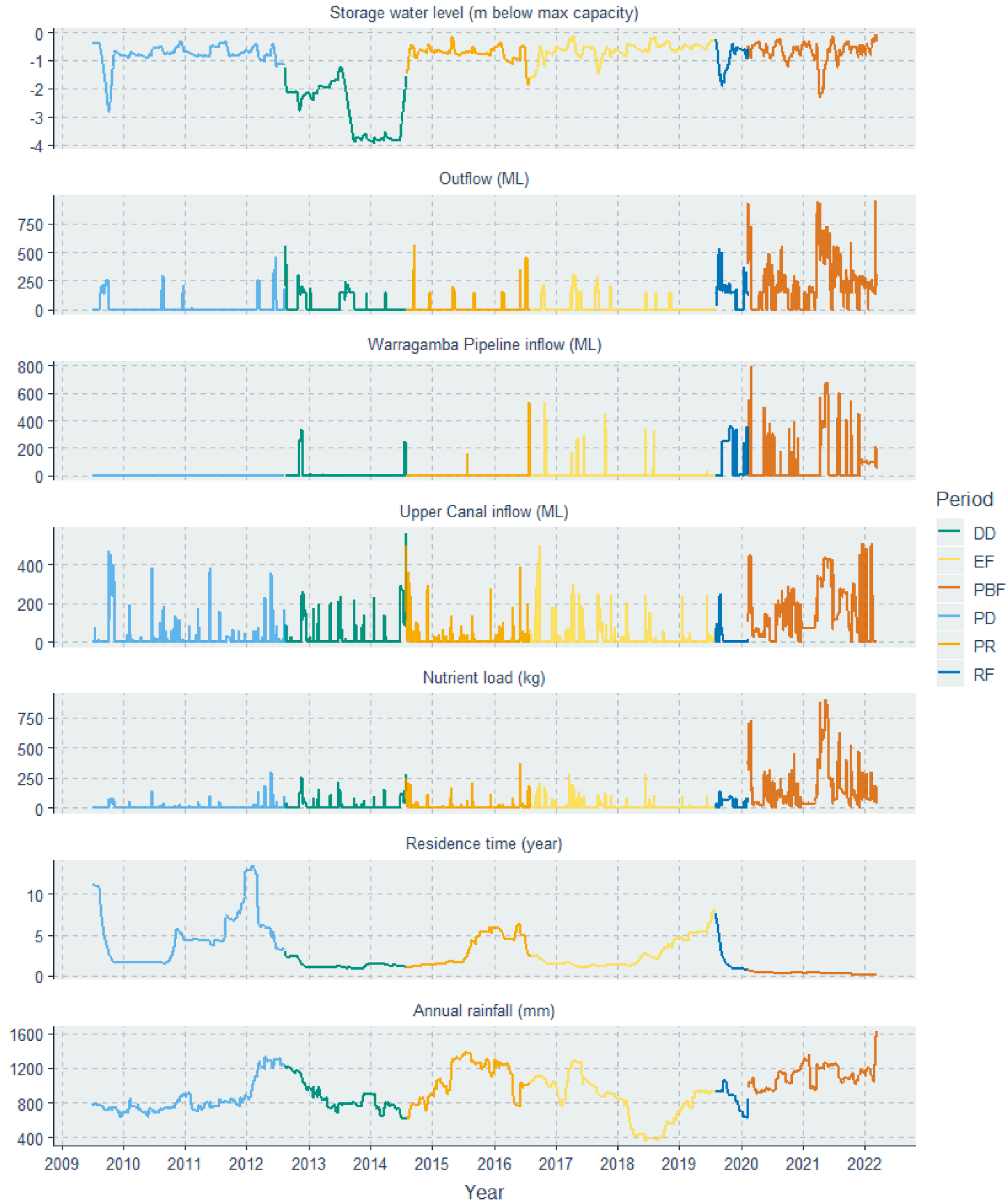

Fig. 2. Time series of hydrological variables at Prospect Reservoir: storage water level, outflow, Upper Canal inflow, Warragamba Pipeline inflow, nutrient load and residence time and annual rainfall. (For interpretation of the references to color in the text, the reader is referred to the web version of this article.).

3.4. Functional groups

During the whole study period, 83 phytoplankton taxa were identified and sorted into 22 functional groups (Table 2). Eleven functional groups were considered dominant since they were present in >50 % of samples and on average each made up more than 2 % of total biovolume: **A, B, D, E, F, J, K, Lo, N, X2, Y** (Table 2). The 11 dominant groups on average accounted for more than 90 % of the total biovolume and thus, were used in later PCA and RDA analysis; group **M** was also included in this analysis, as while it was not considered dominant, it is a group of concern for water management.

In the period leading up to the introduction of more regular flow activity (2009–2019), there were no major seasonal or annual variations in functional group community composition, as illustrated in Figs. 6 and 7. Changes were instead mainly attributed to biovolume fluctuations with higher average biovolume typically observed during the warmer seasons (late-summer and early-autumn). Seasonal phytoplankton biovolume variation also became more pronounced post-drawdown

Table 1
Descriptions of operational/hydrological phases in Prospect Reservoir over the study period (2009–2022).

Period	Date range	Characteristic
Pre-drawdown (PD)	Sep-2009 to Aug-2012	Limited inflow/outflow (flow) activity. Mean residence time of 4.76 ± 0.09 years. Periodic inflow to maintain reservoir at just below storage capacity (0.5 m). Inflow only sourced from Upper Canal.
During drawdown (DD)	Aug-2012 to Jul-2014	Limited flow activity. Mean residence time of 1.3 ± 0.02 years. Inflow is primarily sourced from Upper Canal. Drawdown commissioned for dam-wall maintenance. Two major drawdowns during winter of 2012 and 2013. The event saw an approximate one-third (21,000 ML) and half (16 000 ML) reduction in storage volume respectively. Reservoir maintained at approximately 2 and 4 m below storage level.
Post-refill (PR)	Aug-2014 to Jul-2016	2-year period following refilling. Limited flow activity. Mean residence time of 3.19 ± 0.07 years. Periodic inflow to maintain reservoir at just below storage capacity. Inflow mainly sourced from Upper Canal.
Episodic flow (EF)	Aug-2016 to Jul-2019	Limited flow activity. Mean residence time of 2.59 ± 0.05 years. Periodic inflow to maintain reservoir at just below storage capacity. Inflow mainly sourced from Upper Canal.
Regular flow (RF)	Aug-2019 to Feb-2020	Regular flow activity. Mean residence time of 1.91 ± 0.13 years. Inflow sourced from Warragamba. Storage level mostly maintained at just below storage capacity.
Post-bushfire (PBF)	Mar-2020 to Mar-2022	Regular flow activity. Mean residence time of 0.32 ± 0.004 years. High nutrient loading from inflows due to extreme event impacts. Storage level mostly maintained at just below storage capacity.

(Fig. 6). Following an increase in inflow regularity with good water quality (RF period), biovolume appeared to be relatively low compared to prior years, and the major dominance of **Lo** was replaced with a community with fairly even dominance of several groups including **A, B, F, E** and **K**. Following regular poor water quality inflow (PBF) activity, biovolume increased and the community shifted again, with the reduction/loss of groups **A** and **E**, and an increase in the dominance of groups **B, D** and **F**.

3.5. PCA on functional groups

The PCA using the 11 dominant functional groups and group M explained 72.32 % of community data variability in the first two axes (axis 1 = 55.05 %; axis 2 = 17.32 %) (Fig. 7). The most important group for axis 1 ordination was **Lo** (0.994). For axis 2, the most important groups for its ordination were **B** (0.738), **F** (0.828), **D** (0.646), **J** (0.578), **Y** (0.435) and **M** (0.403). The PCA results indicated three main FG community compositions. The first community (blue ellipse) is located in the negative side of axis 1 and 2 of the PCA and can be described as a less productive (low biovolume) community, mainly dominated/co-dominated by group **A**, occasionally dominated/co-dominated by group **B, E** and **F**, and negatively associated with group **Lo**. This community was found prior to poor-quality inflow and is the only community composition observed during the regular flow (RF) period. The second community (yellow and grey ellipse) is located in the positive side of axis 1, described as a community with **Lo** as the major dominant group, and was observed prior to RF. The PCA also revealed the post-drawdown period (DD, PR, and EF) saw more frequent sampling periods with higher Lo productivity (grey ellipse). The third community (orange ellipse) was the only community observed post-bushfire (PBF), and was positively associated with groups **B, D, F, J, N** and **M** and also distinguished by the loss in dominance and/or absence of groups **A, E** and **Lo**.

3.6. Redundancy analysis

Redundancy analysis (RDA) revealed relationships between environmental variables, functional group biovolume and operational period (Fig. 8). The first two axes accounted for 33.86 % of the variability in functional group biovolume and environmental variables (axis 1: 25.22 %, axis 2: 8.64 %). The forward selection of environmental variables selected five significant variables, Mn ($F = 26$, $p = 0.001$), WT ($F = 11.5$, $p = 0.001$), EC ($F = 10.4$, $p = 0.001$), DO ($F = 6.1$, $p = 0.003$),

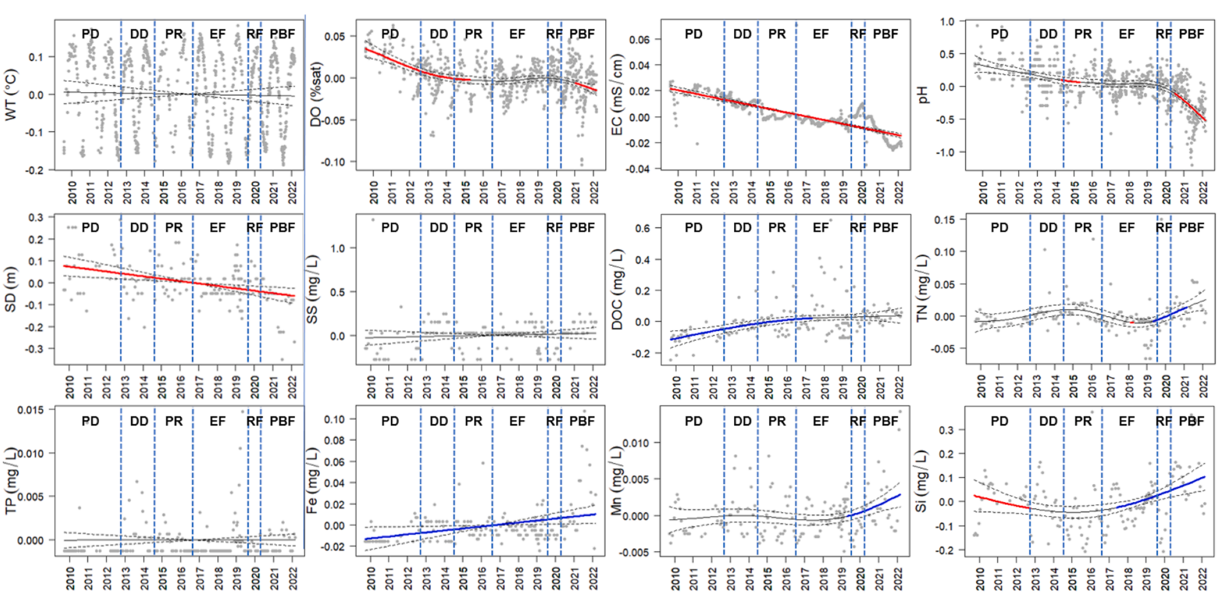

Fig. 3. Time series of water quality variables (log(1 + x) transformed except for pH): Water temperature (WT), dissolved oxygen (DO(%sat)), pH, electrical conductivity (EC), Secchi depth (SD), total nitrogen (TN), total phosphorus (TP), total iron (Fe), total manganese (Mn), reactive silicate (Si) and dissolved organic carbon (DOC). The solid line represents the regression line fitted by generalised additive mixed models (GAMMs), the broken lines denote the approximate 95 % confidence intervals on the fitted function. Red lines indicate a significant decrease and blue lines indicate a significant increase ($p < 0.05$). (For interpretation of the references to color in the text, the reader is referred to the web version of this article.).

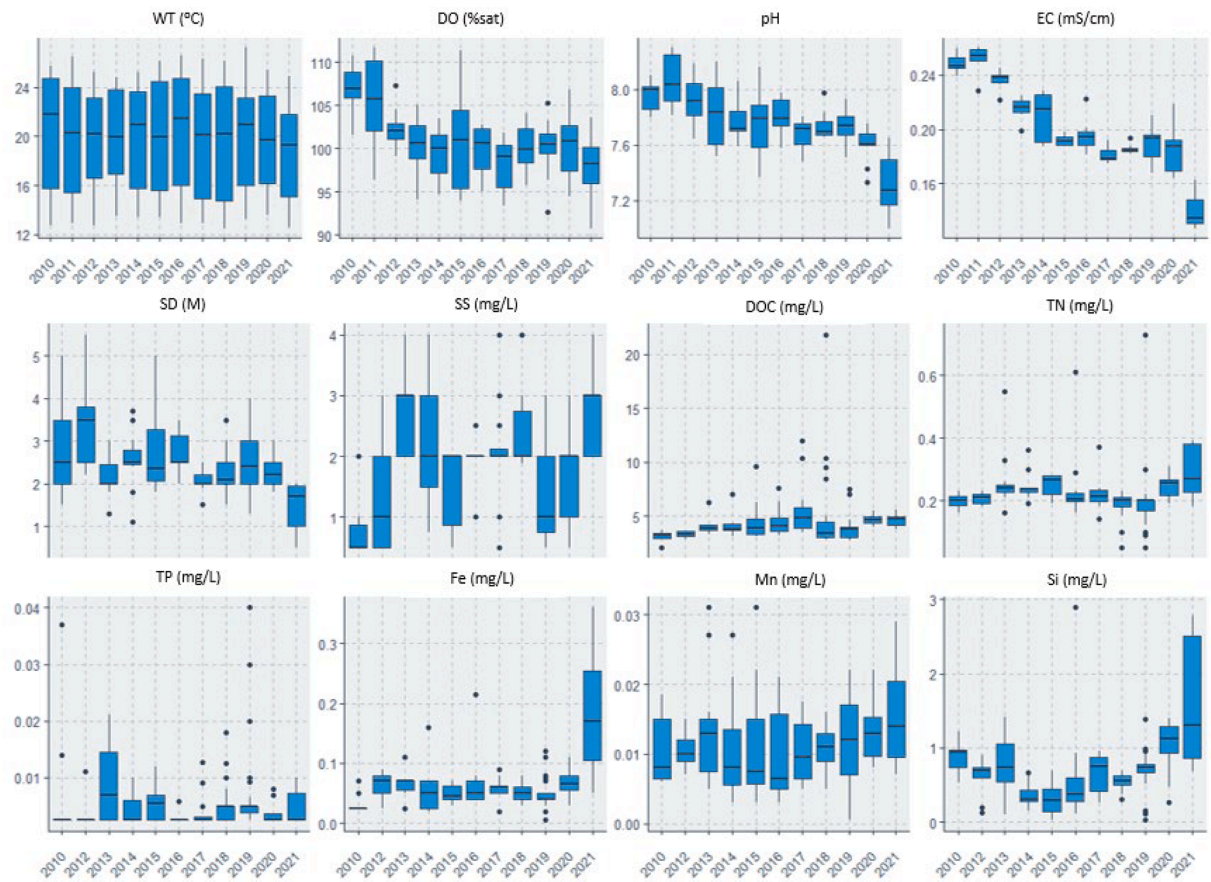

Fig. 4. Annual variation of 12 water quality variables measured at the mid-lake (RPR1) site in Prospect Reservoir: water temperature (WT), dissolved oxygen (DO (% sat)), pH, electrical conductivity (EC), Secchi depth (SD), suspended solids (SS), dissolved organic carbon (DOC), total nitrogen (TN), total phosphorus (TP), total iron (Fe), total manganese (Mn), reactive silicate (Si).

TP ($F = 5.1$, $p = 0.011$), which together accounted for 31.1 % of total variance. The inclusion of all 10 environmental variables accounted for 37.2 % of total variance. The first axis was positively correlated with Mn (0.529), Fe (0.255) and negatively correlated with EC (−0.415), SD (−0.393) and DO (−0.221). The second axis was positively associated with WT (0.237), TP (0.254) and Mn (0.235), and negatively associated with DO (−0.187).

Samples were distributed in the RDA triplot according to the operational period. The first axis separated the samples from the post-bushfire (PBF) period and the preceding periods (PD, DD, PR, EF and RF). The PBF period showed higher nutrient concentrations (Fe, Si, Mn), and low SD and EC, which corresponded with higher productivity (Chl-*a*) and biomass of functional groups including **B, D, F, J** and **M**, and lower biomass in groups **A** and **E**. The second axis mainly separated the pre-drawdown (PD) and regular flow (RF) period from the post-drawdown periods prior to regular flow (DD, PR, and EF). The periods DD, PR, and EF were primarily placed in the positive side, with high TP and WT contributing to high Lo productivity. Group K was also positively related to these periods and was also associated with high Mn. The negative side of axis 1 and 2, were predominantly associated with the PD period, which were characterised by high DO and low nutrients (TP and Mn), contributing to lower phytoplankton biomass overall. The Monte Carlo test showed all canonical axes were significant ($F = 8.5$, $p = 0.001$; 999 random permutations) and explained 39.87 % of total variance.

3.7. Trophic indicators

Chl-*a* had a significant increasing trend during the post-bushfire period (Fig. 9). Between 2010 - 2019, Chl-*a* concentrations indicated mesotrophic conditions (Fig. 10). In 2010 (pre-drawdown), the mean annual Chl-*a* concentration was 2.87 ± 0.33 μg/L, whereas the yearly mean concentration in 2013 (during drawdown) was 4.24 ± 0.36 μg/L. Mean annual Chl-*a* concentrations from 2014 - 2019 ranged from 3.14 – 3.83 μg/L. Chl-*a* concentrations measured in the years following the 2020/2019 bushfires (2020 and 2021) had the highest recorded yearly average values and indicate eutrophic conditions (6.57 ± 0.82 and 7.88 ± 1.24 μg/L respectively).

Q index decreased significantly from mid-2018 to the end of the study period (Fig. 9). The Q index suggested excellent water quality over most of the study period with the average Q index during the summer ranging from 4 - 4.44 from 2010 to 2019 (Fig. 10). This range excludes the summer of 2014, 2016 and 2018, where the average Q index dropped to good quality (3.63 ± 0.16, 3.75 ± 0.15 and 3.86 ± 0.17 respectively). During the post-bushfire period, the average summer Q index consistently stayed within good quality, with values of 3.64 ± 0.26 and 3.88 ± 0.11, for 2020 and 2021, respectively. Notably, 2022 recorded the lowest ever average summer Q index of 3.24 ± 0.07.

4. Discussion

Phytoplankton communities are primarily shaped by resource availability, with nutrients, temperature and light availability being important factors (Facey et al., 2019; Reynolds, 2006). In regulated lake and reservoir systems like Prospect Reservoir, hydrodynamic changes due to management practices can influence these factors and impact the phytoplankton community and water quality. The use of functional grouping proved effective in enhancing our understanding of phytoplankton dynamics in Prospect Reservoir. The 2019/2020 summer

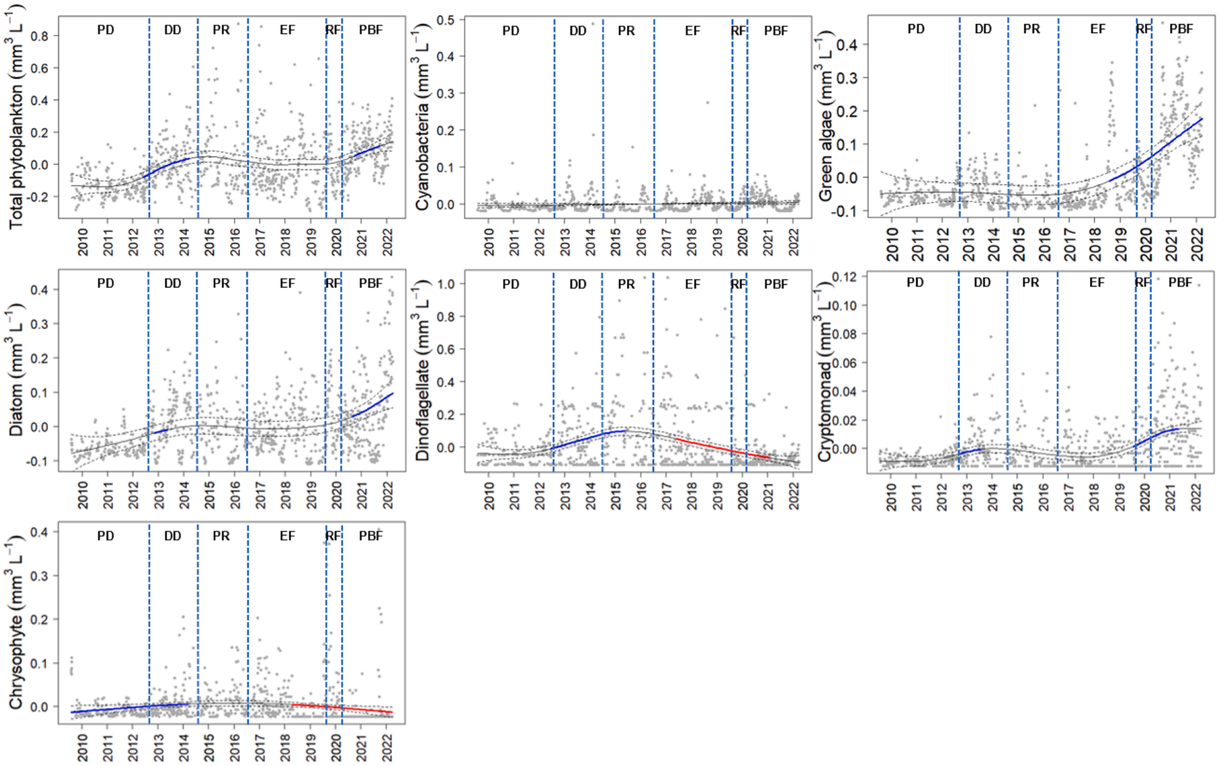

Fig. 5. Time series of $\log_{10}(1 + x)$ transformed biovolume of total phytoplankton, cyanobacteria, green algae, diatom, dinoflagellate, cryptomonad and crysophyte. The solid line represents the regression line fitted by generalised additive mixed models (GAMMs), The broken lines denote the approximate 95 % confidence intervals on the fitted function. Red lines indicate a significant decrease, and the blue lines indicate a significant increase ($p < 0.05$). (For interpretation of the references to color in the text, the reader is referred to the web version of this article.).

bushfire and subsequent heavy flooding that impacted connected reservoirs, was linked to an increase in nutrient loading. These events led to a shift in the phytoplankton community towards functional groups representative of meso-eutrophic conditions (Reynolds et al., 2002), accompanied by reduced water quality indicators. To a lesser degree, an extended drawdown and subsequent refill-event also had a significant effect on phytoplankton dynamics and water quality.

4.1. Drawdown/refill effects

The drawdown event coincided with a significant increase in phytoplankton growth. This was evidenced by an increase in total phytoplankton biovolume associated with an increase in dinoflagellate, diatom, cryptomonad and chrysophyte biovolume. Water quality declined post-drawdown, including increased total iron (Fe) and dissolved organic carbon (DOC), decrease in Secchi depth (SD) and dissolved oxygen (%sat) (DO), and a higher frequency of total phosphorus (TP) levels measured above the LOD (0.005 mg/L). Higher dissolved nutrients could be linked to macrophyte decomposition (Lu et al., 2018; Watts, 2000), as found by Lu et al. (Lu et al., 2018), where drying and rewetting of macrophyte beds resulted in significant phosphorus release. Additionally, sediment disturbance could be another source of phosphorus and iron (Pickering, 1994), potentially associated with wave action induced during water level drawdowns/refills and/or from increased sediment exposure to low water levels (Wildman and Hering, 2011). Higher DOC concentrations are associated with eutrophic systems (Wen et al., 2020), and aligned with (Lewis et al., 2023) findings of elevated DOC linked to phytoplankton bloom degradation following an extended drawdown.

RDA revealed a strong correlation between TP and WT, with large dinoflagellates (**Lo**) linked with the years following the drawdown event (2012–2019). Members of the functional group **Lo** were mainly represented by large dinoflagellates including *Peridinium* spp. and *Ceratium* sp. This group was the most important functional group in Prospect Reservoir for most of the study period, when flow was predominantly offline (2009–2019). Dinoflagellates have the ability to store a surplus of phosphorus, and use it for several generations of subsequent divisions without requiring additional phosphorus supply (Pollingher, 1988; Serruya and Berman, 1975). This stochastic TP environment post-drawdown could explain the strong relationship between TP and **Lo**. Furthermore, high WT is favourable for the growth of this group (Grigorszky et al., 2003; Reynolds et al., 2002; Xiao et al., 2011; Zhu et al., 2013). Warmer periods are typically when the water column is more stable, which dinoflagellates are adapted to, enabling them to utilise their flagella to reach optimal growth conditions.

4.2. Effect of fire inflows

The 2019/2020 summer bushfires burnt a significant portion of Sydney's main water supply catchment (Fig. 1), leading to a notable influx of ash and sediment into the reservoir after rainfall (Neris et al., 2021; Yang et al., 2020). This influx can be linked to the increased loading of nutrients that include TP, TN, Fe and Mn (Sánchez-García et al., 2023). Subsequent heavy flooding events, driven by a transition to a La Niña weather pattern, prolonged poor-quality runoff in the catchment. Consequently, Sydney's main reservoirs experienced an extended period of poor water quality from 2020 to 2022. Water managers responded by diverting poor-quality water into Prospect Reservoir, where it was diluted with cleaner water, supplementing Sydney's water supply with improved water compared to that from the main reservoirs. Post-bushfire, Prospect Reservoir experienced an increase in phytoplankton productivity, indicated by rising trends in Chl-*a* and total phytoplankton biovolume. This coincided with changes in water quality, including increased dissolved nutrients (Mn, Fe, Si, TN), and lower SD

Table 2
Phytoplankton functional groups found in Prospect Reservoir over the whole study period (2009–2022), the corresponding factor F, habitat description, and species represented by each functional group with dominant functional group and species indicated by (α) and (*), respectively (Padisák et al., 2006; Padisák et al., 2009; Reynolds et al., 2002).

Group	F factor	Habitat	Genus/species
A$^{\alpha}$	5	Clear, often well-mixed, base poor lake	*Acanthoceras* sp.; *Rhizosolenia eriensis; Thalassiosira* sp.; *Urosolenia* sp.*
B$^{\alpha}$	3	Vertically mixed, mesotrophic small-medium lake	*Aulacoseira* sp.*; *Cyclotella* sp.*
D$^{\alpha}$	2	Shallow, enriched turbid waters	*Encyonema* sp.; *Nitzschia* sp.; *Synedra* sp.*
E$^{\alpha}$	2	Usually small, oligotrophic, base poor lake	Not identified Chrysophyte; *Dinobryon* sp.*; *Mallomonas* sp.
F$^{\alpha}$	5	Clear, deeply mixed meso-eutrophic lake	*Ankistrodesmus* spp.*; *Dictyosphaerium* sp.; *Kirchneriella* sp.; *Oocystis* sp.*; *Nephrocytium* sp.; *Sphaerocystis* sp.
H1	1	Eutrophic, both stratified and shallow lakes with low nitrogen	*Chrysosporum* sp.; *Chrysosporum bergii; Dolichospermum* spp.; *Nostocaceae*
J$^{\alpha}$	1	Shallow, enriched lake	*Crucigenia* sp.; *Coelastrum* sp.; *Golenkinia* sp.; *Pediastrum* sp.; *Scenedesmus* spp.*; *Tetraedron* sp.; *Tetrastrum* sp.
K$^{\alpha}$	2	Shallow, nutrient-rich water column	*Aphanocapsa* sp.*; *Aphanothece* sp.; *Cyanocatena* sp.; *Cyanodictyon* sp.; *Cyanogranis* sp.; *Cyanonephron* sp.*; *Rhabdogloea* sp.
L$_{\mathbf{O}}^{\alpha}$	5	Deep and shallow, oligo-eutrophic, medium-large lake	*Ceratium* spp.*; *Peridinium* spp. *
M	0	Eu-hypertrophic, small-medium lake	*Microcystis* spp.
MP	5	Frequently stirred up, inorganically turbid shallow lake	*Achnanthes* sp.; *Cymbella* sp.; *Eunotia* sp.; *Navicula* sp.
N$^{\alpha}$	5	Mesotrophic epilimnia	*Cosmarium* sp.; *Pleurotaenium* sp.; *Staurastrum* sp.; *Staurodesmus* spp.; *Spondylosium* sp.*; *Tabellaria* sp.; *Pleurotaenium* sp.
P	5	Eutrophic epilimnia	*Fragilaria* sp.; *Melosira* sp.
S1	0	Turbid mixed environments. Only includes shade-adapted cyanobacteria	*Planktolyngbya* sp.; *Planktothrix* sp.; *Pseudanabaena* sp.
T	5	Deep, well-mixed epilimnia	*Mougeotia* sp.; *Planctonema* sp.;
W1	0	Small organic ponds	*Euglena* sp.
W2	0	Small, clear mixed layers, meso-eutrophic lakes	*Trachelomonas* sp.
W$_{\mathbf{S}}$	0	Ponds rich in organic matter from decomposition of vegetal matter (humic environments)	*Synura* sp.
X1	4	Shallow, eu-hypertrophic lake	*Ankyra* sp.; *Monoraphidium* sp.
X2$^{\alpha}$	3.5	Shallow, meso-eutrophic lake	*Carteria* sp.; *Chroomonas* sp.*; *Chrysochromulina* sp.
X3	4	Shallow, well mixed oligotrophic lake	*Chlamydomonas* sp.; *Schroederia* sp.
Y$^{\alpha}$	2	Range of habitats where representative species live in almost all lentic when grazing pressure is low	*Cryptomonas* sp.*; *Gymnodinium* sp.;

and DO, which are markers of eutrophication (ANZECC, 2000).

Post-bushfire, functional groups **B, D, F, N**, and **J** increased, representing diatoms and green algae typical of meso-eutrophic systems (Padisák et al., 2009; Reynolds et al., 2002), while dinoflagellates (**Lo**) and chrysophytes (**E**) decreased. Increased dominance of green algae and shifts in diatom species, especially the prominence of *Synedra* sp. (**D**), can be indicators of eutrophication (Reynolds et al., 2002). Furthermore, the present study aligns with Saad et al.'s (Saad et al., 2016) findings of a shift in mixotrophic taxa along an increasing trophic state gradient, whereby primarily-heterotrophic mixotrophs (chrysophytes) are replaced by primarily-autotrophic mixotrophs (cryptomonads). Unlike algae with flagella or buoyancy regulation, diatoms and green algae rely on water column mixing for survival due to their high sinking rate. Groups **B, D** and **F** are known to dominate in well-mixed waterbodies (Padisák et al., 2009). Increasing flow can enhance water column mixing (Feipeng et al., 2013; Xiao et al., 2011) and may explain the dominance of green algae and diatoms over cyanobacteria and dinoflagellates following regular inflows. Liuxihe Reservoir, an oligo-mesotrophic reservoir in China, had similar dominant functional groups to Prospect Reservoir (**A, B, E, F, Lo, X2**) (Xiao et al., 2011). Xiao et al. (Xiao et al., 2011) found that water column instability, linked to high precipitation during the monsoon season, was associated with the loss in dominance of group Lo. Zhou et al. (Zhou et al., 2021) observed that diatoms are better adapted to fluctuating light levels compared to dinoflagellates. This may be relevant in the context of the post-bushfire period where inflows increased and light availability decreased (measured as SD).

This present study suggests the importance of Mn to phytoplankton productivity (Chl-*a*) in Prospect Reservoir, with Mn and Chl-*a* showing the strongest positive correlation. Group **K**, characterised by picoplankton including *Aphanocapsa* sp. and *Cyanonephron* sp., that typically dominate in eutrophic conditions (Padisák et al., 2009) made up the majority of the cyanobacteria community in Prospect Reservoir. Cyanobacteria had a strong seasonal trend, with higher biovolumes during warmer periods, and RDA revealed a positive relationship with WT. This supports extensive evidence for cyanobacterial preference for the warmer seasons (O'Neil et al., 2012). However, cyanobacteria (**K**) had the strongest relationship with Mn, and higher cyanobacterial biovolume was also observed during the cold season in 2018, that coincided with Mn concentrations at comparatively higher levels. Manganese is an important trace metal for the healthy functioning of most organisms, and like iron, is essential for photosynthetic processes (Facey et al., 2019; Facey et al., 2022). While evidence of trace metal limitation in waterbodies has increased (Facey et al., 2021), particularly iron limitation (Xing and Liu, 2011), this is presently the only study that suggests manganese significantly affects phytoplankton dynamics in a lake/reservoir. This may be due to manganese normally being more abundant in most aquatic systems (Facey et al., 2019). The mobilisation of trace metals from connected catchments, triggered by extreme events, may have alleviated trace metal limitations, thereby enhancing phytoplankton productivity (Chl-*a*). While the increase in Mn levels post-bushfire can be linked to allochthonous sources, the reasons behind the pronounced seasonality of Mn, particularly its elevated concentrations during the cold season of 2018, remain unclear. Manganese typically releases from sediments during periods of stratification (Pickering, 1994), but such stratification has not been observed in Prospect Reservoir. This discrepancy points to the potential existence of unexplored internal dynamics influencing Mn levels, which merits further investigation.

Higher nutrient availability is often associated with increased cyanobacterial dominance in aquatic environments (Smith et al., 1999). However, during the post-bushfire period cyanobacterial biovolume did not significantly increase. *Microcystis* (**M**) was virtually absent in Prospect Reservoir before the post-bushfire period. Its subsequent presence, aligned with higher counts in Warragamba (WaterNSW 2021; WaterNSW 2022), indicating that the donor reservoir may be a potential source of the nuisance taxa (Fornarelli and Antenucci, 2011). However, while the abundance of *Microcystis* did increase, it remained well below guideline levels used by water managers (WaterNSW 2021; WaterNSW 2022). This suggests that seeding events and nutrient enrichment (O'Neil et al., 2012), may not be sufficient disturbances to trigger the

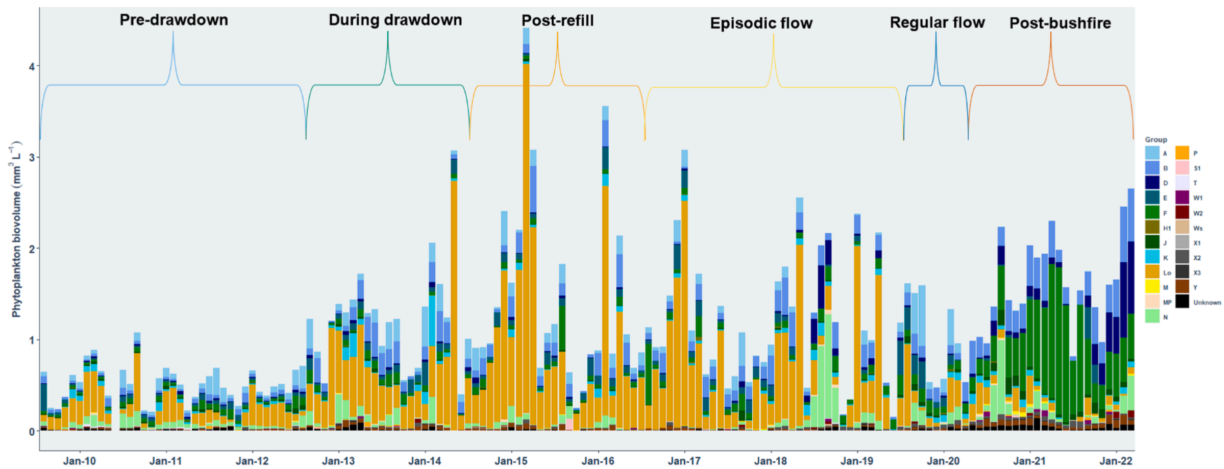

Fig. 6. Mean monthly biovolumes (mm^{3}L^{-1}) of phytoplankton functional groups in Prospect Reservoir during the study period (2009–2022). (For interpretation of the references to color in the text, the reader is referred to the web version of this article.).

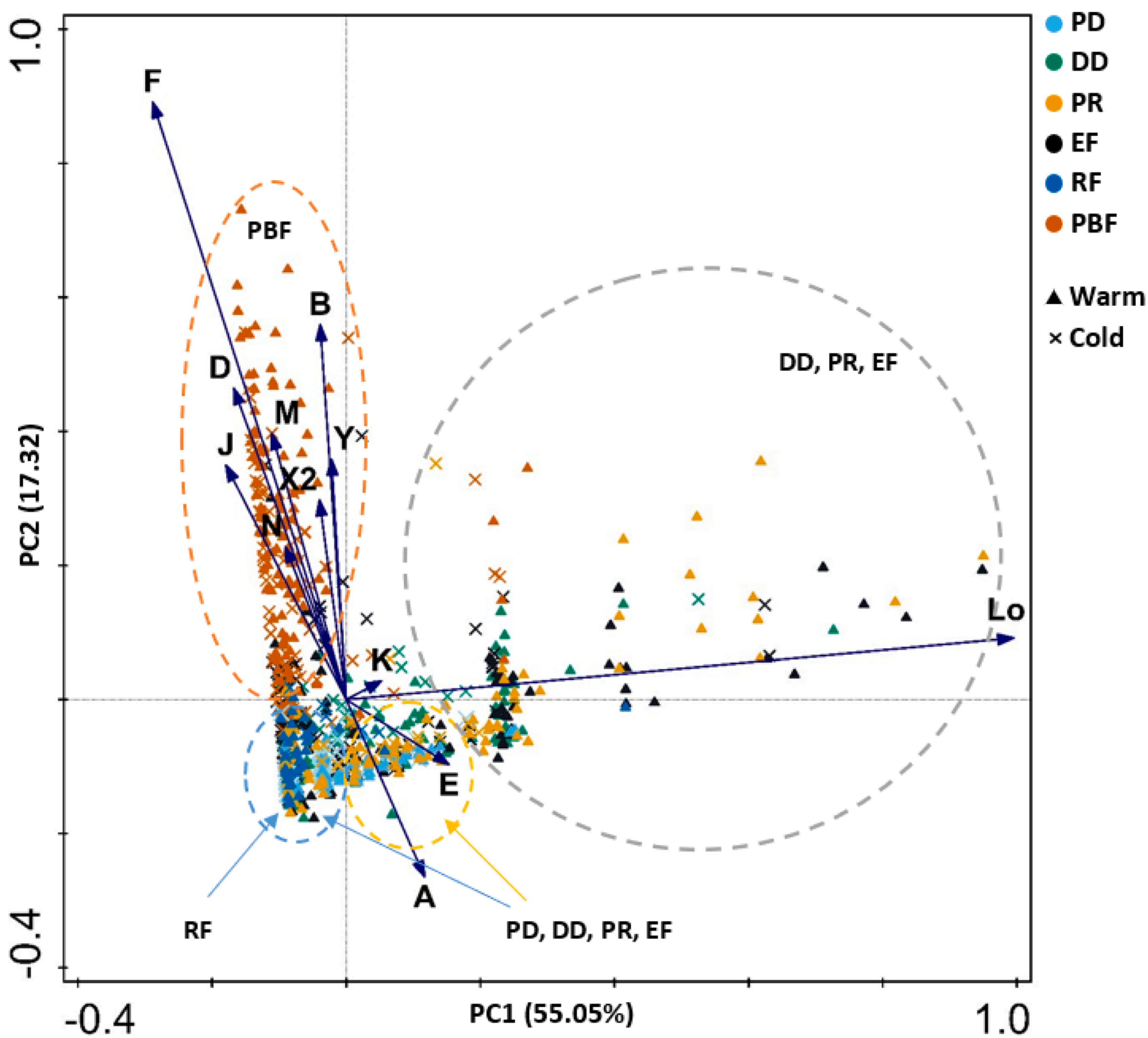

Fig. 7. Principal component analysis (PCA) applied to functional group community data from station RPR1 in Prospect Reservoir over the whole study period (2009–2022). Sampling units distinguished operational period and season. Operational period: post-drawdown (PD) light-blue; during-drawdown (DD) = dark-green; post-refill (PR) = yellow; episodic flow (EF) = light-yellow; regular flow (RF) = dark-blue; post-bushfire (PBF) = dark-orange. Seasons: warm = triangle; cold = cross. Functional groups included in the ordination comprise of 11 dominant groups (A, B, D, E, F, J, K, Lo, N, X2, Y) and group M. (For interpretation of the references to color in the text, the reader is referred to the web version of this article.).

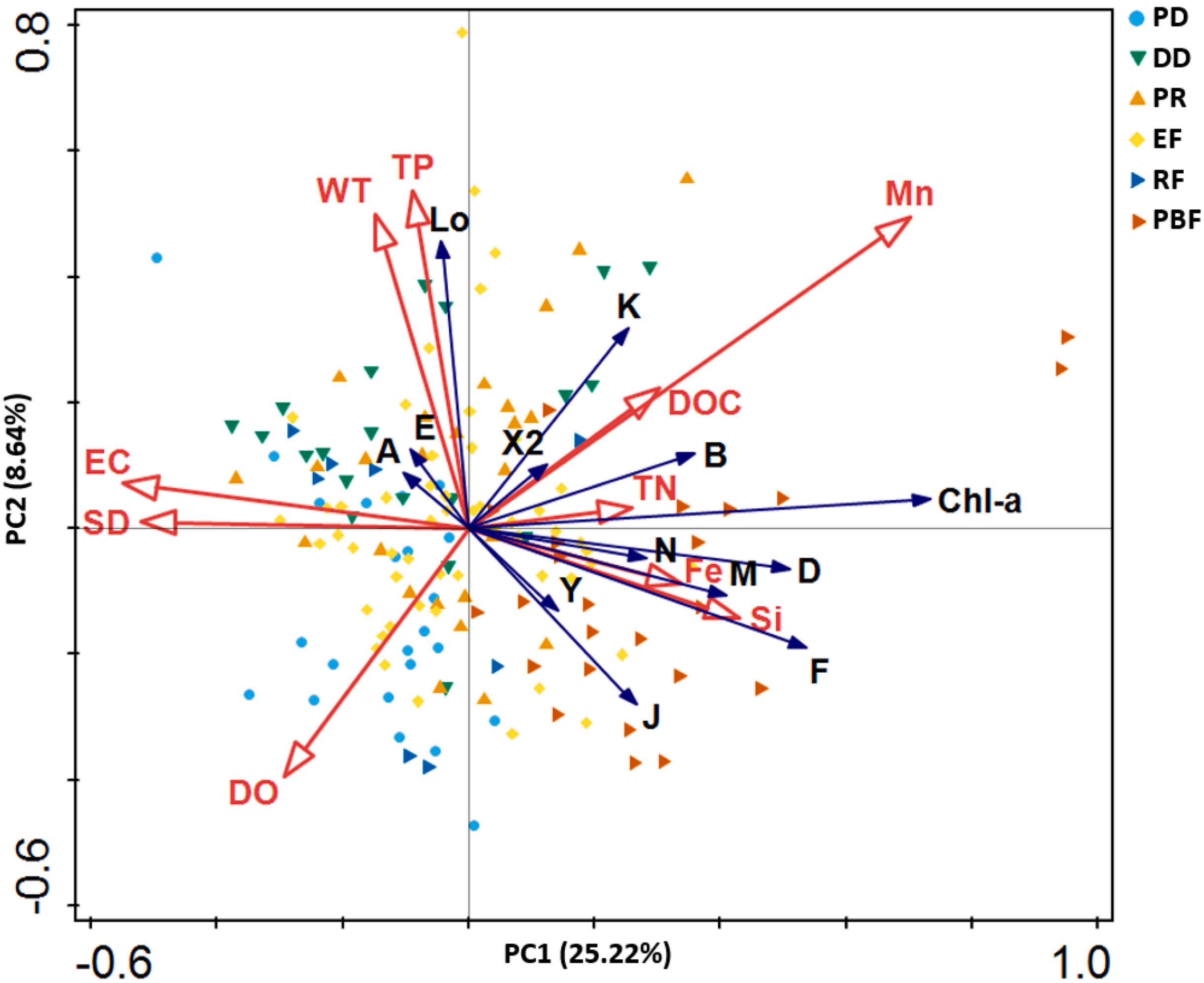

Fig. 8. Triplot diagram for RDA of data from station RPR1 (mid-lake) in Prospect Reservoir. Environmental variables: water temperature (WT), dissolved oxygen (DO), electrical conductivity (EC), Secchi depth (SD), dissolved organic carbon (DOC), total nitrogen (TN), total phosphorus (TP), total manganese (Mn), total iron (Fe), reactive silicate (Si). Chlorophyll-*a* (Chl-*a*) , group M and dominant functional groups: A, B, D, E, F, J, K, Lo, N, X2, Y. Samples labelled in relations to operational period: post-drawdown (PD) = blue circle; during-drawdown (DD) = dark-green down-triangle; post-refill (PR) = yellow up-triangle; episodic flow (EF) = light-yellow diamond; regular flow (RF) = dark-blue right-pointed-triangle; post-bushfire (PBF) = dark-orange right-pointed-triangle. (For interpretation of the references to color in the text, the reader is referred to the web version of this article.).

dominance of cyanobacteria over other algal groups in Prospect Reservoir at its current state. Rather, specific favourable conditions, like a stable water column that enhances cyanobacteria's light access through buoyancy, might be integral for their growth (Mitrovic et al., 2001). A 2011 study on water transfers within the Shoalhaven scheme (Fornarelli and Antenucci, 2011), reported similar findings to our present study. They observed increased phytoplankton productivity (Chl-*a*) and iron levels following high-water transfers, along with higher total phytoplankton biovolume, associated with diatoms and cryptomonads. Notably, they found lower cyanobacteria levels during high flows, particularly a decrease in *Microcystis*. Therefore, continuously operating aerators in Prospect Reservoir and reducing water residence time could be factors preventing nuisance cyanobacteria from dominating (Visser et al., 2016).

4.3. Reservoir management implications

In NSW, trophic status assessments using Chl-*a* with ANZECC and ARMCANZ (ANZECC, 2000) guidelines, suggest a threshold of 5 μg/L for drinking water reservoirs. Prospect Reservoir, previously mesotrophic, would be reclassified as eutrophic in 2020 (post-bushfire) after Chl-*a* levels surpassed this limit. However, WaterNSW (WaterNSW 2021) reports indicated no significant algal or water quality issues despite these elevated Chl-*a* levels, questioning the adequacy of this sole indicator for effective water quality assessment. To offer a more nuanced understanding of ecological health, this study also incorporated the functional groups assemblage index (Q index). Results indicated 'excellent' water quality for most of the study, with a post-bushfire deterioration to 'good' quality marked by an increase in *Synedra* (group **D**), a species known to dominate in shallow, turbid and eutrophic environments (Padisák et al., 2009). While cell concentrations observed in this study are not presently concerning, this species can complicate water treatment as it can clog filters and potentially produce taste and odour compounds (Rose et al., 2019). Adopting the OECD's (Bowles, 1982) recommendation of an 8 μg/L threshold for mesotrophic conditions may be a more relevant benchmark for managing this reservoir as it would better align with the Q index results and abiotic indicators which also did not exceed ANZECC and ARMCANZ guidelines (ANZECC, 2000).

Despite notable biotic changes indicative of eutrophication, positive TP trends were not observed post-bushfire. Prospect Reservoir's TP levels were often below the limit of detection (<0.05 mg/L), indicating oligotrophic conditions (ANZECC, 2000), a contrasting finding to the usual Chl-*a* and TP association commonly observed in waterbodies

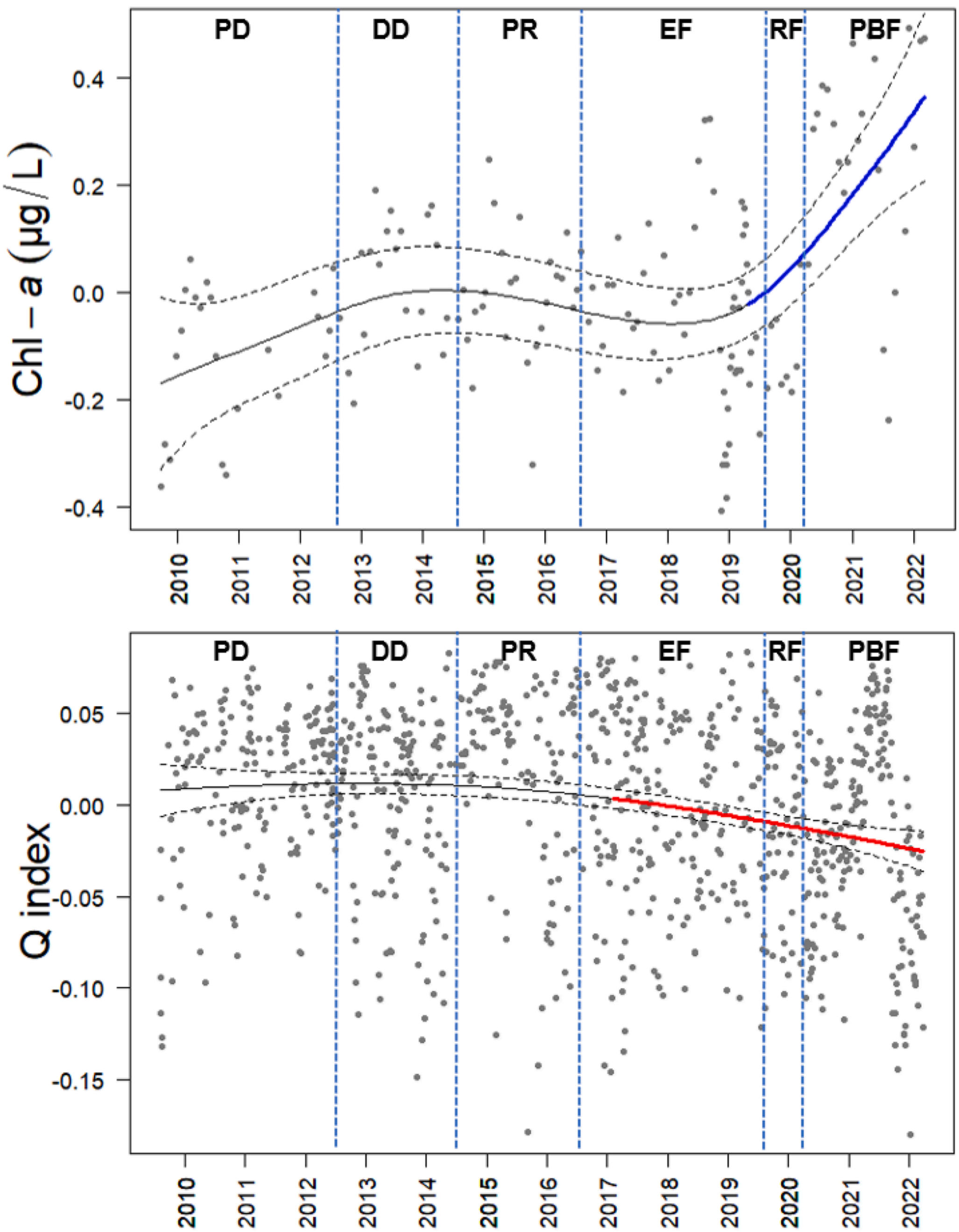

Fig. 9. Time series of $\log_{10}(1 + x)$ transformed trophic indicators: Chlorophyll-a (Chl-a) and Functional group assemblage index (Q index). The solid line represents the regression line fitted by generalised additive mixed models (GAMMs), The broken lines denote the approximate 95 % confidence intervals on the fitted function. Red indicate a significant decrease, and blue lines indicate a significant increase ($p < 0.05$). (For interpretation of the references to color in the text, the reader is referred to the web version of this article.).

(Carvalho et al., 2013). The absence of a Chl-*a* and TP relationship could be attributed to inadequate sampling frequency, whereby a lag in response between nutrient input and phytoplankton productivity is not effectively captured (Strobl and Robillard, 2008). Additionally, biological factors may play a role, as illustrated by the case of Ninféias Pond in Brazil, where high Chl-a levels occurred despite low TP, linked to the presence of macrophyte beds (Fonseca and Bicudo, 2010; Fonseca and Bicudo, 2011). The reservoir's historical alum treatment for phosphorus sequestration could also play a role, though the expected longevity of alum's effectiveness has likely elapsed (Welch and Cooke, 1999; Welch et al., 1994).

Reducing water residence time has been shown to improve water quality (Lucas et al., 2009; Mitrovic et al., 2001; Padovesi-Fonseca et al., 2009; Woo et al., 2021), but this depends on the comparative quality of the recipient and incoming water. The Sydney region's main water reserves are more productive than Prospect Reservoir, indicating that frequent inflow redirection could exacerbate eutrophication, a concern supported by findings from Hamilton and Schladow (Hamilton and Schladow, 1995) and observations from this study. Climate change compounds this challenge, as demonstrated by the 2019/2020 summer bushfires, the worst wildfire on record that impacted a key Sydney water supply catchment, leading to extended water quality deterioration. This highlights the need for proactive measures, such as increased prescribed burns, to prevent such extreme wildfires, which have demonstrated the most severe and prolonged impacts (Paul et al., 2022). Moreover, given the established importance of macrophytes in promoting 'clear-water'

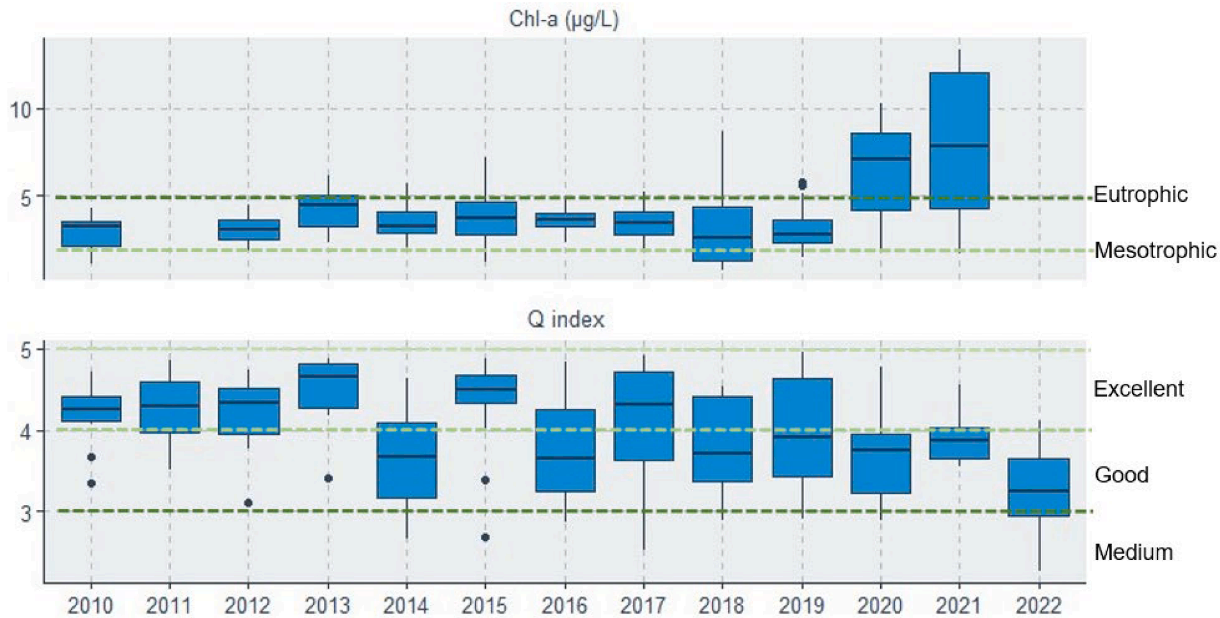

Fig. 10. Annual variation of trophic indicators at the mid-lake site (RPR1) at Prospect Reservoir: chlorophyll-a (Chl-a) and functional group assemblage index (Q index).

states in other systems (Hilt, 2015; Hilt et al., 2010; Scheffer et al., 1993; Song et al., 2019), it is reasonable to consider their potential role in Prospect Reservoir. Understanding whether macrophytes can similarly benefit Prospect Reservoir is crucial to sustainable water management, particularly considering the water quality risks of eutrophication associated with their die-off.

5. Conclusion

Hydrological changes from flow management significantly impacted water quality and phytoplankton dynamics in Prospect Reservoir. Factors such as TP and water temperature emerged as important drivers shaping the phytoplankton community, particularly following an extended drawdown and refill event. The findings indicate a trophic state shift, notably triggered by an increase in poor-quality inflows from connected reservoirs, that was linked to the 2019/2020 summer bushfires and subsequent heavy rainfall events. This shift was evidenced by elevated total biovolume, Chl-*a*, and dissolved nutrient levels (including TN, Mn, Fe, Si), alongside diminished levels of dissolved oxygen and SD. Phytoplankton Functional group dynamics also shifted, marked by a decline in oligotrophic groups (**A, E, Lo**) and a corresponding rise in meso-eutrophic groups (**B, D, F, J, N, M**). Moreover, the study highlights the significant role of total manganese, and its positive association with Chl-*a* and cyanobacterial biovolume, a finding not yet observed in any other waterbody. Using Q index, Prospect Reservoir transitioned from ‘excellent’ to ‘good’ water quality during the lowest drawdown period and the increase in nutrient loading post-bushfire. These findings highlight the utility of integrating functional grouping and Q index for effective water quality assessment. This study also identifies the risk of eutrophication of pristine water supplies due to increased flow activity, a concern further compounded by extreme events including bushfires and successive heavy flooding.

CRediT authorship contribution statement

Huy A. Luong: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Validation, Visualization, Writing – original draft, Writing – review & editing. **Ann-Marie Rohlfs:** Data curation, Funding acquisition, Supervision, Validation, Writing – review & editing, Project administration, Resources. **Jordan A. Facey:** Conceptualization, Supervision, Writing – review & editing. **Anne Colville:** Conceptualization, Supervision, Writing – review & editing. **Simon M. Mitrovic:** Conceptualization, Funding acquisition, Resources, Supervision, Validation, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper. Huy A. Luong reports financial support was provided by WaterNSW. Huy A. Luong reports financial support was provided by Commonwealth of Australia. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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