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Drivers of water quality in Okawa Bay, Lake Rotoiti



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Summary

Okawa Bay is a shallow, sheltered embayment at the western end of Lake Rotoiti that has experienced poor water quality and cyanobacterial blooms for many decades. This report brings together historical information, long-term monitoring, and recent high-frequency measurements to better understand what controls water quality in the bay, how conditions have changed over time, and the influence of management activities.

Water quality has improved in some respects since the early 2000s, including reductions in nutrient and chlorophyll-*a* concentrations. However, these improvements have been modest, and cyanobacterial blooms have become more frequent again since 2021. Compared with other monitored locations in Lake Rotoiti, Okawa Bay continues to have relatively high algal biomass, lower water clarity, and a greater occurrence of potentially toxic cyanobacteria.

Poor water quality in Okawa Bay reflects a combination of its shallow, sheltered setting, historical nutrient enrichment, weather and climate, and exchange of water with the wider lake. Calm conditions can result in temporary stratification and low oxygen near the lakebed, potentially releasing nutrients stored in the sediments. Wind can then mix these nutrients back into the water column and also affect water exchange with the western basin. These processes can occur over periods of days and are not always captured by routine monthly monitoring.

Management actions have reduced some external pressures on Okawa Bay, but the long-term record does not show a single sustained improvement that can be attributed to one intervention. Similarly, while aquatic weed spraying can release nutrients as plants decay, observed changes following spraying were generally within normal seasonal variability. Overall, Okawa Bay remains particularly susceptible to poor water quality because of its physical setting and legacy of historical nutrient enrichment. Continued management of external nutrient inputs remains important, while higher-frequency monitoring would help better understand the short-lived processes that contribute to poor water quality and blooms.

Technical Summary

This report examines long-term water quality, cyanobacterial blooms, and the factors driving water quality in Okawa Bay, a shallow embayment (max depth ~5 m) at the western end of Lake Rotoiti. The assessment combines historical information, long-term water quality monitoring, and high-frequency temperature and dissolved oxygen measurements collected during summer 2025–2026. These data were used to examine changes in water quality over time, compare water quality in Okawa Bay with that at four other long-term water quality monitoring sites across Lake Rotoiti, and investigate the potential influence of climatic conditions, water exchange, nutrient availability, and management activities.

This assessment addressed four objectives in relation to water quality: documenting recent patterns, comparing Okawa Bay with other Lake Rotoiti sites, evaluating key drivers, and assessing evidence for management-related signals, including aquatic weed control and the Ōhau Channel diversion.

Historical evidence indicates that Okawa Bay has exhibited recurrent summer stratification, oxygen depletion, and algal bloom development since at least the early 1970s. Internal nutrient loading was estimated in the early 2000s to substantially exceed external catchment inputs during summer months. These historical estimates reflect legacy nutrient enrichment and sediment accumulation within the bay. Previously reported residence times of 7–12 days under calm conditions (reducing to ~3.7 days under wind forcing) indicate strong sensitivity to wind.

Routine monitoring by the Bay of Plenty Regional Council from 2003–2025 shows modest long-term improvement in chlorophyll-a (chl-a) and nutrient concentrations, although Secchi depth has declined overall since 2007. Breakpoint analysis detected temporal shifts in chl-a (2006), Secchi depth (2007, 2010, 2016), dissolved reactive phosphorus (DRP; 2011), and dissolved inorganic nitrogen to DRP ratio (DIN:DRP; 2022), but no single intervention was associated with a persistent step change across all variables at monthly resolution. DIN:DRP ratios have increased in recent years, indicating that dissolved inorganic nitrogen has become more abundant relative to phosphorus since 2022, although the mean ratio remains within the potential co-limitation range, consistent with persistently high total nitrogen to total phosphorus (TN:TP) ratios and suggesting an increasing relative importance of phosphorus concentrations in regulating bloom development.

Since 2021, cyanobacterial bloom frequency has increased across multiple Lake Rotoiti sites, including Okawa Bay, relative to the lower bloom frequency observed from the mid-2000s to late-2010s. Although some improvements in nutrient concentrations and chl-a are evident relative to the early 2000s, the magnitude of change is modest, and continued

bloom susceptibility is associated with the bay's shallow morphology and accumulated sediment nutrient stores.

High-frequency logger data confirm that Okawa Bay is intermittently stratified (polymictic), with frequent mixing events and strong diurnal variability. Stratification events co-occur with declines in bottom water dissolved oxygen concentrations. Monthly monitoring likely underestimates the frequency and duration of short-lived anoxic events, and subsequent potential for episodic internal nutrient loading. Differences between surface and bottom nutrients from monthly observations were generally small, consistent with frequent mixing, but episodic bottom-water enrichment was observed. Although bottom-water nutrients were not significant predictors of chl-a at monthly resolution, high-frequency data indicate conditions conducive to episodic internal nutrient release that may not be captured by routine sampling.

Relative to other Lake Rotoiti sites, Okawa Bay exhibits higher mean chl-a (12.0 μg L^{-1}), lower mean Secchi depth (3.2 m), and the highest median toxic cyanobacteria biovolume. Since cyanobacteria monitoring began in 1996, 41 bloom events, defined as exceedances of the 1.8 mm^3/L potentially toxic cyanobacterial biovolume threshold, have been recorded in Okawa Bay.

Quantile regression and moving window analysis indicate that water quality in Okawa Bay is primarily influenced by meteorology (windspeed, rain), climate (Southern Oscillation Index), and nutrient conditions at Site 3 in the western basin of Lake Rotoiti, immediately outside of Okawa Bay. Water transport between Site 3 and Okawa Bay is mediated by suitably aligned winds (NE-SW), and seasonal meteorological patterns. The strength of relationships between Okawa Bay water quality and both Ōhau Channel nutrient loads and Site 3 nutrient concentrations varied through time. Relationships between nutrient inputs from Lake Rotorua and Okawa Bay water quality were generally weak before construction of the Ōhau Channel diversion wall in 2008 and remained variable after its construction, making it difficult to determine the effects of the diversion wall on Okawa Bay water quality from these statistical relationships alone.

To assess the potential effects of aquatic weed spraying, nutrient release from decaying macrophytes was modelled following recorded spray events and examined alongside observed changes in whole-bay nutrient mass. The modelling indicates that sprayed biomass has the potential to release nitrogen and phosphorus. However, monthly monitoring did not reveal consistent, sustained increases in whole-bay nutrient mass following spraying. Given the relatively short water-exchange timescales in Okawa Bay, these observations cannot resolve short-lived nutrient release or its potential ecological effects.

Overall, the findings indicate that Okawa Bay functions as a shallow, meteorologically sensitive embayment with ongoing nutrient legacy effects and moderate but persistent hydrodynamic connectivity to the western basin. Bloom dynamics reflect the interaction between nutrient releases from sediments and meteorological regulation of mixing and residence time, with climatic variability also influencing bloom conditions. Management interventions appear to have reduced some external nutrient loading but have not fundamentally altered the physical–ecological dynamics governing bloom susceptibility. Okawa Bay therefore remains inherently bloom-prone due to its shallow polymictic structure and sediment nutrient stores, suggesting that continued management of external nutrient inputs remains important, although substantial improvements may be constrained by persistent internal nutrient cycling and hydrodynamic processes.

1. Introduction

1.1 Background and purpose of this study

The Bay of Plenty Regional Council (BOPRC) is responsible for the ongoing monitoring and management of water quality within Lake Rotoiti, including Okawa Bay. While lake-wide management interventions over the past two decades have contributed to improvements in some water quality indicators, spatial heterogeneity across Lake Rotoiti remains evident. In particular, Okawa Bay continues to exhibit recurrent cyanobacterial blooms and distinct water quality behaviour relative to other monitored sites in Lake Rotoiti.

Over recent decades, multiple management interventions have been implemented within the Rotorua–Rotoiti system to reduce nutrient inputs and mitigate bloom risk. These include the reticulation of wastewater systems within the Okawa Bay catchment, diversion of Rotorua-derived inflows via the Ōhau Channel diversion wall (completed in 2008), alum dosing in Lake Rotorua to reduce bioavailable phosphorus, and aquatic macrophyte control within Okawa Bay. Aquatic macrophyte spraying, in particular, has been undertaken periodically in Okawa Bay as part of an operational weed management programme aimed at controlling invasive submerged vegetation, including species such as *Egeria densa*, *Lagarosiphon major*, and hornwort (*Ceratophyllum demersum*). Treatments have typically occurred during late spring to early summer, when plant biomass is high and access to the bay is important for recreational and cultural use. The programme has varied in spatial extent between years, with treated areas ranging from small, targeted zones to more extensive coverage of littoral beds. While spraying is intended to manage nuisance vegetation and maintain navigability, collapse and decomposition of treated biomass may release nutrients to the water column. Understanding whether such release produces detectable changes in whole-bay nutrient dynamics is therefore an important component of evaluating the ecological implications of the weed management programme.

In recent years, bloom occurrences within Okawa Bay have raised questions regarding the dominant drivers of phytoplankton variability, the relative influence of climatic and hydrodynamic factors, and the potential contribution of local management activities, including aquatic weed control and the diversion of Ōhau Channel inflows from Lake Rotorua. Although long-term monthly monitoring data exist for a range of physicochemical and biological variables, these datasets have not previously been synthesised in a comprehensive, driver-focused assessment specific to Okawa Bay.

To address this knowledge gap, BOPRC commissioned the University of Waikato to undertake a structured investigation into the patterns and drivers of water quality and cyanobacterial bloom dynamics in Okawa Bay. The purpose of this work is thus to provide BOPRC with a data-driven assessment of the factors influencing water quality (inclusive

of cyanobacterial bloom behaviour) in Okawa Bay, within the context of broader Lake Rotoiti processes and management interventions. Specifically, BOPRC sought to:

- Document patterns in water quality using the most recent data from Okawa Bay
- Determine whether water quality dynamics in Okawa Bay differ from sites in the greater Lake Rotoiti basin
- Evaluate the relative roles of climatic variability, meteorological forcing, hydrological exchange, and internal nutrient processes on water quality dynamics in Okawa Bay
- Examine whether available data provide evidence for management-related signals, including aquatic weed control and the Ōhau Channel diversion

To address these key objectives, we separated our analysis into four components:

1. Examining observed patterns in long-term monitoring records of water quality in Okawa Bay to describe the basic limnology of the bay (shown in Results 3.1)
2. Comparing long-term water quality and cyanobacterial dynamics in Okawa Bay with sites across the greater Lake Rotoiti basin (Results 3.2)
3. Analysing the main drivers of water quality and factors which mediate these relationships in Okawa Bay (Results 3.3)
4. Quantifying whether relationships between water quality and drivers have changed over time in response to management actions (Results 3.4)

1.2 Limnological description of historical literature on Okawa Bay

1.2.1 Overview

Okawa Bay is a small, shallow (maximum depth ≈ 5.0 m) embayment on the southwest end of Lake Rotoiti. Literature describing Okawa Bay often demonstrates that the bay has physical and hydrodynamic characteristics which are distinct from the main Lake Rotoiti basin (e.g., Gibbs et al. 2003). Okawa Bay exhibits several characteristics which are especially unique: (i) a shallow morphology and sheltered setting, (ii) strong sensitivity to short-lived stratification and subsequent bottom-water oxygen depletion, which then mobilises nutrients from bottom sediments, (iii) long hydraulic residence time and (iv) restricted water connectivity with the main Lake Rotoiti basin due to a narrow, shallow inlet to the bay. In a National Institute of Water and Atmospheric Research (NIWA, Note NIWA merged into Earth Sciences New Zealand in July 2025) synthesis prepared in 2003 to evaluate engineering options for Lake Rotoiti and the Ōhau Channel, Okawa Bay is described as “small” and “shallow” and noted for calm conditions; it also records historic (1930s) use of the bay as a timber-milling landing place and the likelihood that sawdust was dumped there episodically (Gibbs, et al. 2003). This, among more recent land use pressures, have likely had large and potentially long-lived contributions to the pool of sediment nutrient and organic matter. Decades of organic matter deposition likely

remain in the sediment pool with dramatic, lasting influence on internal loading and oxygen depletion dynamics in the shallow bay.

Despite these unique characteristics, Okawa Bay has received relatively little attention in research studies conducted across the Rotorua lakes. Below, we summarise available literature to help contextualise the historical understanding of the limnology of Okawa Bay.

1.2.2 Intermittent stratification, oxygen loss, and bloom response, 1970s

The combination of sheltered and shallow depths in Okawa Bay can still produce stratification under calm conditions. In the NIWA engineering assessment, Gibbs et al. (2003) cite a 1972 study (Fish & Bryers, 1973) undertaken specifically to evaluate whether diverting the Ōhau Channel through Okawa Bay could improve water quality via enhanced flushing. The same NIWA report summarises that monthly observations in 1972 showed frequent summer oxygen loss in the lower water column, with a minimum recorded dissolved oxygen of 6 mg/L, implying stratification, and that this event was followed by an algal bloom of about 50 mg/m^3 chl-a (Gibbs et al., 2003). This early evidence has been repeatedly invoked in later reports to characterise the dynamics of Okawa Bay, despite limited subsequent verification of stratification events and dissolved oxygen dynamics.

1.2.3 Macrophyte regime change, 2000s

By the early 2000s, long-term ecological transitions were being documented in Okawa Bay. Ray (2001) described the bay as having originally supported extensive native charophyte beds that stabilised sediments and contributed to clear-water conditions. During the mid-1980s, however, macrophyte beds declined substantially following invasion by exotic species such as hornwort, and the system shifted toward algal dominance. Increased nutrient concentrations in the water column were identified as a likely contributing factor, and subsequent years were characterised by incomplete macrophyte recovery and recurrent algal blooms (Ray, 2001; Scholes, 2015a).

In response to growing concern about bloom frequency and intensity, Rotorua District Council requested advice on whether sewerage reticulation would produce measurable water-quality improvements. Ray (2001) addressed this question directly by quantifying plausible external nutrient inputs. Household septic tanks and sewage contributions from a nearby resort lodge were estimated to contribute up to approximately 1.6 tonnes of nitrogen per annum to Okawa Bay, representing roughly two-thirds of the external nitrogen load to the bay at that time. Locally-derived catchment inputs were comparatively small, consistent with the small contributing area and lack of permanent surface inflows. Ray (2001) also emphasised that geothermal springs, if present beneath the bay, could supply potentially significant amounts of additional ammonia-rich nitrogen. However, the existence and magnitude of such geothermal inputs was not quantified.

Importantly, Ray (2001) cautioned that removal of septic inputs alone might not yield rapid improvement because sediments had accumulated nutrients over many years of macrophyte and algal decay. He explicitly noted that sediment nitrogen could continue to supply the water column for years or decades, unless flushing increased sufficiently to export nutrients more rapidly via transport pathways.

1.2.4. Internal loading estimates, residence time, and intensified bloom biomass, 2002–2003

Early process-based investigations (Ray et al., 2002; Gibbs et al., 2003) provided the first quantitative assessments of nutrient dynamics in Okawa Bay. Benthic chamber experiments undertaken by Ray et al. (2002) estimated annualised sediment nutrient releases of 17–77 kg N day^{-1} (as NH_4–N) and 0.7–9.3 kg P day^{-1} (as DRP). These fluxes exceeded estimated catchment and septic nitrogen inputs by approximately 4–19 fold, leading to the conclusion that internal loading dominated short-term nutrient supply to the bay. Gibbs et al. (2003) further interpreted these findings within the broader hydrodynamic context of Lake Rotoiti, emphasising the importance of exchange flow between Okawa Bay and the western basin and the role of intermittent stratification in enhancing sediment nutrient release.

Nutrient–phytoplankton modelling indicated that removal of septic inputs and diversion of 1 m^3 s^{-1} of Ōhau Channel water would reduce typical summer algal biomass by only ~15% over decadal time scales (<10 years). The limited predicted response reflected (i) dominant sediment nutrient supply and (ii) substantial water exchange between Okawa Bay and the western basin of Lake Rotoiti (mean ~5 m^3 s^{-1}), which reduced the proportional influence of local load reductions. These early analyses therefore described Okawa Bay as a system constrained by internal nutrient legacies and water exchange with the western basin of Lake Rotoiti rather than by contemporary external loading alone.

Gibbs et al. (2003) also linked the internal loads to the bay's sheltered position and limited exchange with the western basin. Hydrodynamic modelling reported by Gibbs et al. (2003) indicated that water-exchange times of approximately 7–12 days under calm conditions, decreasing to about 3.7 days when wind-driven circulation increased exchange with the western basin. The longer exchange times provide sufficient opportunity for substantial algal biomass accumulation, particularly where sediment-derived nutrients are available. Conversely, when wind forcing enhances horizontal exchange and vertical mixing, reduced residence time and increased dilution would be expected to limit net biomass accumulation or disperse developing blooms. The authors therefore did not characterise Okawa Bay as permanently isolated, but as a system whose bloom expression is sensitive to the interaction between meteorological forcing, thermal structure, and sediment nutrient release (Gibbs et al., 2003; Hamilton et al., 2009a, b).

Gibbs et al. (2003) also compared oxygen depletion and bloom intensity across decades, stating that oxygen depletion events occurred again in 2002 and 2003, including at least one period of anoxia, and that subsequent bloom biomass was 3–5 times higher than in 1972.

1.2.5 Toxic cyanobacteria and microcystin concentrations, 2003–2004

Wood (2004) documented bloom-forming and toxic cyanobacteria in New Zealand lakes, including Okawa Bay. During February 2004, microcystin concentrations in Okawa Bay reached 410 µg L^{-1}, associated with *Microcystis aeruginosa* cell densities of approximately 940,000 mL^{-1}. Microcystins remained detectable until the conclusion of sampling in May 2004 (Wood, 2004).

1.2.6 Hydrodynamics and water exchange with Lake Rotoiti, 2004-2010

Between 2004 and 2010, a series of field and modelling investigations clarified the nature of hydrodynamic connectivity between Okawa Bay and the western basin of Lake Rotoiti, particularly in the context of Ōhau Channel diversion and Okere Gate operations. These studies characterised exchange as constrained, density-structured, and temporally variable.

Hamilton et al. (2009b) focused primarily on basin-scale hydrodynamics and the behaviour of Rotorua-derived inflows entering Lake Rotoiti. BioFish transects demonstrated strong vertical stratification in the main basin and clear horizontal gradients in temperature and conductivity between Okawa Bay and the adjacent western basin. These gradients were interpreted as evidence of reduced horizontal mixing through the shallow connecting channel under certain conditions. However, Hamilton et al. (2009b) did not calculate a specific residence time for Okawa Bay. Instead, they used conductivity differences of approximately 35 µS/cm between Lake Rotorua and Lake Rotoiti as conservative tracers to identify intrusion pathways. Their analysis showed that Rotorua-derived inflows frequently mostly plunged to form interflows or underflows rather than mixing uniformly at the surface, with intrusion depth controlled by density structure and influenced by wind forcing. The authors concluded that transport within Lake Rotoiti is strongly density-structured and meteorologically sensitive, implying that the degree of coupling between Okawa Bay and the western basin varies with stratification and wind regime.

Three-dimensional modelling by Muraoka et al. (2010), using ELCOM–CAEDYM, further quantified the variability of inflow influence at a central location within Okawa Bay. Conservative tracer simulations across multiple years showed pronounced seasonal oscillation in tracer concentration at the bay, increasing from January through April and declining by July. Peak tracer concentrations were higher in wetter years (1998 and 2001) and lowest in a dry year (2000), demonstrating that hydrological forcing measurably alters inflow penetration into Okawa Bay. Von Westernhagen (2010) evaluated multiple operational scenarios for the Ōhau Channel diversion wall and quantified how these

scenarios altered the tracer fraction at Okawa Bay relative to a defined baseline configuration. Under diversion-wall operational scenarios, the modelled change in tracer fraction at the Okawa Bay station was typically between approximately –3% and +4% relative to the baseline diversion case. In contrast, when comparing pre-diversion conditions to diversion configurations, changes in tracer fraction at the site were roughly –10% to +20% in some simulations.

1.2.7 Management responses, 2010s

Following sewerage reticulation in Okawa Bay (completed in 2006), monitoring data did not show a general improvement in water quality between approximately 2005 and 2010 (Bay of Plenty Regional Council data). However, in 2011, monitoring data showed increases in both nitrogen and phosphorus concentrations, declines in Secchi depth, and elevated chl-a relative to preceding years (McIntosh, 2015). That year, an aerial application of approximately 5 tonnes of AqualP™ (a modified zeolite clay mineral) was undertaken in January to suppress a developing algal bloom. While bloom biomass was reduced below recreational guideline thresholds following treatment (McIntosh, 2015), this reduction appeared to have been short lived (Scholes, 2015a)

The 2011 deterioration in water quality was interpreted in monitoring reports as potentially arising from multiple interacting mechanisms, including nutrient release associated with decaying macrophytes following weed spraying, sediment nutrient release during stratification and oxygen depletion, and degraded inflow conditions linked to diversion-wall and outlet operations (McIntosh, 2015; Scholes, 2015a). Rather than attributing the bloom to a single cause, these reports emphasised the interaction of management actions and internal nutrient cycling.

In February 2014, 5 tonnes of alum was applied in response to a bloom event. Technical assessment indicated that approximately 8 tonnes would have been required to fully bind the dissolved reactive phosphorus present at that time, implying that the applied dose was insufficient to permanently suppress phosphorus availability under the prevailing conditions (Scholes, 2015a).

Trend analyses of cyanobacteria across Rotorua lakes further indicate interannual variability in bloom magnitude, but persistent spatial heterogeneity, with Okawa Bay exhibiting higher bloom potential than more open basin locations (Wood et al., 2022). Monitoring and intervention records from the 2010s indicate that while reductions in external nutrient loading contributed to Lake Rotoiti water quality improvements, internal nutrient cycling and periods of reduced water exchange between Okawa Bay and the western basin continued to favour recurrent bloom development within the bay (McIntosh, 2015; Scholes, 2015b).

2. Methodology

Data analysis to investigate patterns and drivers of water quality and cyanobacterial bloom dynamics in Okawa Bay were carried out using the statistical software, RStudio (R Core Team 2024). Code and data to reproduce these analyses are available upon request.

2.1 Long-term monitoring data on water quality from BOPRC

Data from 10 sites within Lake Rotoiti, the Ōhau Channel, and Okere Arm of the Lake Rotoiti Kaituna outlet were obtained for this study from BOPRC (Figure 1). We used data from routine monthly water quality monitoring has occurred at Okawa Bay, Site 3, Site 4 in Lake Rotoiti and the Ōhau Channel, as the outflow from Lake Rotorua and main inflow into Lake Rotoiti. In addition, we used data from the long-term, routine cyanobacterial monitoring program which occurs weekly from ~November to May at shoreline sites at Okawa Bay, Okere Arm (the outflow from Lake Rotoiti), Te Weta Bay, Otaramarae, and Hinehopu. All variables used in this report, from long-term monitoring at within-lake sites, nearshore cyanobacteria data, and discharge at the Ōhau Channel, are described in Table 1.

Nutrient loads from the Ōhau Channel were calculated by multiplying the nutrient concentrations measured at the Ōhau Channel long-term monitoring site by daily mean discharge from the same day measured at the Ōhau Channel discharge gauging site at SH33 (Figure 1). When direct measurements of TN data were unavailable, it was estimated as the sum of total Kjeldahl nitrogen and nitrate + nitrite (TKN and NNN) (2001-2009) as these variables are broadly representative of total nitrogen availability despite minor differences in analytical methodology. Data which were compared with long-term monitoring water quality data were subset to the timeframe of the water quality data (2003-2025). To estimate dissolved inorganic nitrogen (DIN), NH_4 and NNN were summed together.

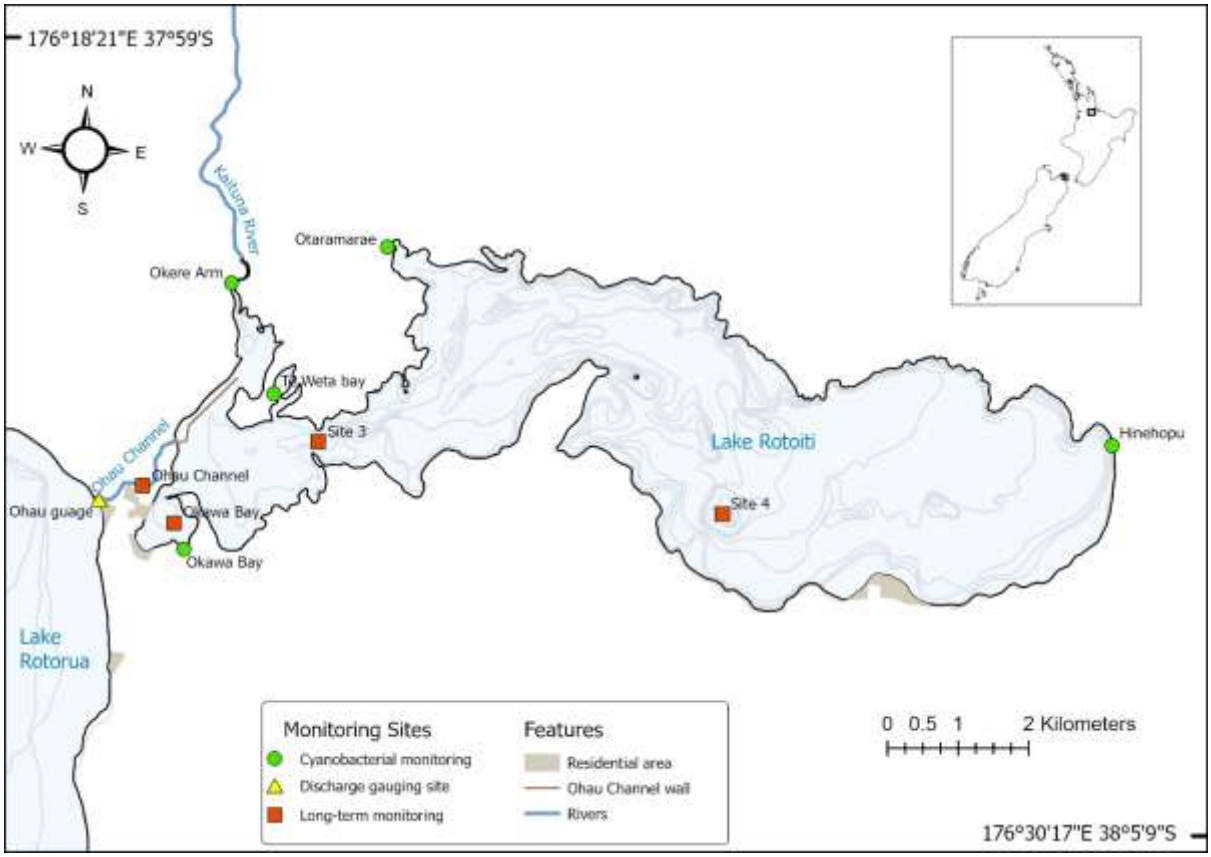


Figure 1. Map of study locations in Lake Rotoiti.

Table 1. Description of water quality data and associated drivers used in this report. Bolded variables are used as responses for water quality analyses. Other variables are used as drivers in water quality analyses or as comparison with Okawa Bay data.

Variable name (units)	**Abbreviation**	**Depth(s)**	**Site(s)**	**Temporal Frequency**	**Temporal Duration**
Water temperature (ºC)	--	Continuous profiles, surface to bottom	Okawa Bay	Monthly	2003-2025
		~every meter	Okawa Bay	10 minutes	18 Dec 2025 – 28 Jan 2026
Dissolved oxygen (mg/L)	DO	Continuous profiles, surface to bottom	Okawa Bay	Monthly	2003-2025
		~every meter	Okawa Bay	10 minutes	18 Dec 2025 – 28 Jan 2026
Chlorophyll-a (µg/L)	Chl-a	Surface integrated	Okawa Bay, Site 3	Monthly	2003-2025
Dissolved reactive phosphorus (µg/L)	DRP	Surface integrated, bottom discrete	Okawa Bay, Site 3, Site 4	Monthly	2003-2025
Ammonium, dissolved	NH_4	Surface integrated, bottom discrete	Okawa Bay, Site 3, Site 4	Monthly	2003-2025

inorganic nitrogen (µg/L)					
Nitrate + nitrite, dissolved inorganic nitrogen (µg/L)	NNN	Surface integrated, bottom discrete	Okawa Bay, Site 3, Site 4	Monthly	2003-2025
Secchi depth (m)	Secchi	--	Okawa Bay, Site 3, Site 4	Monthly	2003-2025
Total nitrogen (µg/L)	TN	Surface	Ōhau Channel, Okawa Bay, Site 3, Site 4	Monthly	2009-2025
Total Kjeldahl Nitrogen	TKN	Surface	Ōhau Channel, Okawa Bay, Site 3, Site 4	Monthly	2003-2012
Total phosphorus (µg/L)	TP	Surface	Ōhau Channel, Okawa Bay, Site 3, Site 4	Monthly	2001-2025
Discharge (m^3/s)	--	--	Ōhau Channel	Daily (mean)	2001-2025
Southern Oscillation Index	SOI	--	--	Monthly	2003-2025
Air temperature (ºC)	--	--	--	Daily (mean)	2003-2025
Precipitation (mm/day)	Rain	--	--	Daily (sum)	2003-2025
Windspeed (m/s)	--	--	--	Daily (median) from ERA5 (Hersbach et al. 2020)	2003-2025
Wind direction	Northeast (NE), Southeast (SE), Northwest (NW), Southwest (SW)	--	Site 3	Daily wind direction and velocity from Limnotrack buoy near Site 3	2013-2020
Toxic cyanobacteria biovolume (mm^3/L)	Toxic BV	Nearshore surface	Okawa Bay, Te Weta Bay, Okere Arm at Kaituna, Hinehopu, Otaramarae	Weekly during summer	1996-2025
Macrophyte spray data	--	--	Okawa Bay	Mostly annual based on operation spray records	2010-2025

2.2 Short-term deployment of high-frequency data in Okawa Bay

In addition to the long-term monthly monitoring data collected by the BOPRC, we deployed a series of high-frequency (every ten minutes) HOBO temperature loggers and a miniDOT dissolved oxygen logger in Okawa Bay as a pilot study to better understand the mixing dynamics in Okawa Bay. We deployed five temperature loggers at 0.5, 1.0, 2.0, 3.0, and 4.0 m in addition to a dissolved oxygen logger a meter above the bottom of the bay (~4.0 m) from December 18, 2025 to January 28, 2026. From this data we calculated thermocline depth using the '*rLakeAnalyzer*' package (Winslow et al. 2019) where the bay was considered stratified if the temperature difference between the top and bottom was greater than 2 ºC. This threshold was used to characterise short-term thermal structure and not as a formal classification of mixing regime.

2.3 Breakpoint analysis

We conducted a statistical breakpoint analysis on key water quality variables to identify potential structural changes in the time series. The purpose of this analysis was exploratory, allowing the data to indicate periods where statistically significant shifts in mean and/or variance occurred, rather than pre-defining change points based on known management interventions or external events. Significant breakpoints therefore indicate time periods where the statistical properties of the monthly observations changed, but do not in themselves imply causation. This analysis was conducted using the *breakpoints()* function in the '*strucchange*' package (Zeileis et al. 2002).

2.4 Seasonal analysis of water quality across sites in Lake Rotoiti

We examine differences in distributions of water quality variables across sites in Lake Rotoiti, as well as across seasons. Summer was defined as December to February; autumn was defined as March to May; winter was defined as June to August; and spring was defined as September to November.

2.5 Frequency, magnitude, and duration of cyanobacterial blooms

To analyse patterns in cyanobacterial bloom dynamics over time, we quantified the frequency, magnitude and duration of blooms across the four cyanobacterial monitoring locations within Lake Rotoiti, as well as the outflow from Lake Rotoiti, Okere Arm from 1996-2025 (Figure 1). We used data on potentially toxin-producing species as this is most relevant for beach closures related to health concerns. If the biovolume of toxic cyanobacteria exceeded the health threshold of 1.8 mm^3/L in a given week, the event was considered a bloom. We then calculated the frequency (number of blooms per year), magnitude (maximum biovolume observed during a bloom event), and duration (number of days where the bloom was observed). These characteristics were calculated each year for each site.

2.6 Quantile regression

One method for assessing the strength and nature of relationships between water quality and drivers is to use quantile regression. This method quantifies the strength of the relationship between a stressor (e.g., external nutrient loads) and a response (e.g., chl-a) at different quantiles within the full distribution of the response variable. For example, this differentiates in the strength of the relationship when the response variable (Okawa Bay chl-a) has high concentrations (e.g., 90th quantile) vs when concentrations are low (e.g., 25th quantile).

We used quantile regression to determine if relationships between surface chl-a in Okawa Bay and various explanatory variables changed at different magnitudes. We examined quantile regression in external weather variables (mean air temperature, sum of precipitation, median windspeed, and the mean SOI over the previous month), internal nutrient sources (bottom water DRP, NH_4-N, and DIN in Okawa Bay, as well as surface DRP, NH_4-N, and chl-a from Site 3), and external nutrient loads (TN and TP from the Ōhau Channel). We examined the 10%, 25%, 50%, 75%, and 90% quantiles between surface chl-a and each stressor. At each quantile, we examined whether the stressor was significant.

We used the package '*quantreg*' (Koenker 2024) to conduct this analysis.

2.7 Assessing the effects of weed spraying on nutrient dynamics

To evaluate whether aquatic macrophyte spraying produced detectable changes in whole-bay nutrient dynamics, an event-aligned analysis was implemented following the conceptual assumptions outlined in Scholes (2015a). An event-aligned quasi-experimental framework was implemented using a macrophyte decay model (Text A1). Spray years included 2010, 2012–2018, 2020, 2022, and 2024. Non-spray years (2019, 2021, 2023) were treated as seasonal controls, with 1 December used as a pseudo-event alignment date. For all events, a 60-day pre-event window was used to define baseline nutrient mass, and a 120-day post-event window was used to evaluate change (Figure 2).

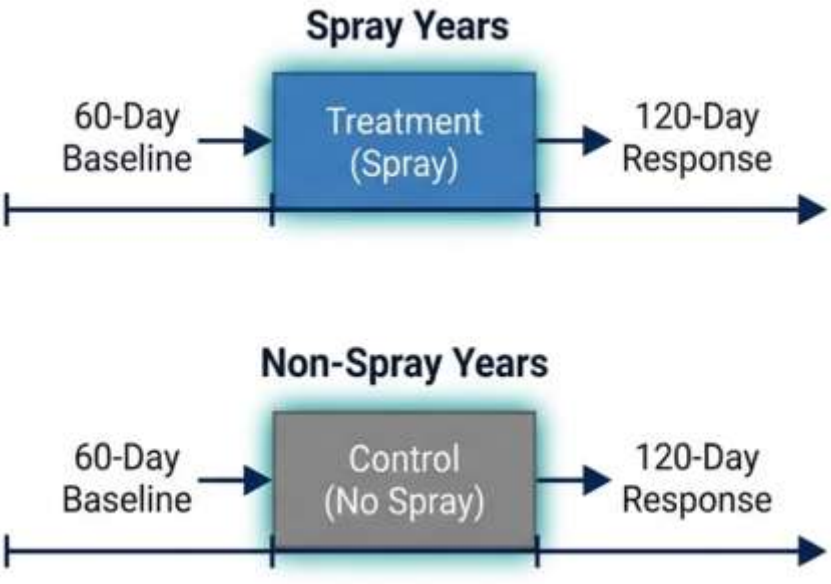


Figure 2. Conceptual overview of the event-aligned weed decay analysis framework applied to Okawa Bay. The treatment (and pseudo-treatment in the control) time period was one day.

Surface-integrated TP and TN concentrations were converted to whole-bay mass (kg) using an estimated bay volume of 1.6×10^6 m^3. Post-event change in mass (ΔM) was calculated relative to the 60-day pre-event mean.

To estimate expected nutrient release from sprayed macrophytes, a first-order decay model was parameterised using the ranges adopted by Scholes (2015a). Treated area was obtained from operational records. Biomass density, tissue nutrient content, and release fraction were sampled from uniform distributions constrained to literature-supported ranges consistent with Scholes (2015a). The macrophyte decay constant was defined such that 95% of releasable biomass decomposed within 84 days. Parameters values for the model are presented in Table 2 and model equations are presented in Appendix A.

For each spray year, 10,000 Monte Carlo simulations were performed to propagate uncertainty in biomass density, tissue nutrient content, and release fraction. Median predicted release and 5th–95th percentile envelopes were calculated.

To compare model predictions with observations, linear rates of change (kg/day) were estimated over the 120 day post-event window for both observed mass and modelled cumulative release. Slopes were compared directly to assess mechanistic adequacy. Spray-year slopes were compared to non-spray slopes using bootstrap resampling to estimate uncertainty in group-level differences.

Table 2. Parameters used in the macrophyte decay model. Full model description can be found in Appendix A.

Parameter	Symbol	Range	Units	Distribution	Source or justification
Sprayed area	A	Spray event specific (1.5-12.22)	ha	Fixed	Operational spray records (Bay of Plenty Regional Council)
Biomass density	B	2,940–4,940	kg dry mass/ha	Uniform	Duarte et al. (1986); Madsen (1993); Scholes (2015a)
Phosphorus content	C_P	3-10	g P kg^{-1} dry mass	Uniform	Barko & Smart (1980); Duarte (1992); Scholes (2015a)
Nitrogen content	C_N	18-36	g N kg^{-1} dry mass	Uniform	Barko & Smart (1980); Duarte (1992); Scholes (2015a)
Decay function	k	–ln(0.05)/84	day^{-1}	Fixed	Carpenter (1980); Webster & Benfield (1986); Scholes (2015a)
Monte Carlo iterations	n	10,000	-	Fixed	Standard Monte Carlo uncertainty propagation practice

3. Results

To assess and describe the general limnological patterns of Okawa Bay, we focused our analysis on five key water quality variables from 2000 to 2025: surface integrated chl-a, as an indicator of phytoplankton productivity; Secchi depth, as a measure of water clarity; surface integrated DRP and DIN, which represent nutrient availability for phytoplankton uptake; and the ratio of DIN:DRP, representing the immediately bioavailable nutrient pool; and the ratio of DIN:DRP, used here as a screening indicator of relative nutrient availability. Lower DIN:DRP values may indicate a greater likelihood of nitrogen limitation in the short term. Although additional water quality variables are monitored by BOPRC, we focus on these parameters as integrative indicators of eutrophication processes and primary productivity in the bay.

3.1 Observed patterns in water quality in Okawa Bay

3.1.1 Long-term patterns in routine water quality monitoring data at Okawa Bay

Significant breakpoints were detected in all variables except DIN (Figure 3; Appendix Table A1).

Surface-integrated chl-a exhibited a breakpoint in 2006. The mean concentration decreased from 22.97 µg/L (2000–2006) to 10.1 µg/L (post-2006). However, this apparent shift was strongly influenced by extreme chl-a values observed in 2003, which included observations up to 234 µg/L. This suggests that the detected breakpoint reflects the influence of early high-magnitude events rather than a sustained structural shift in phytoplankton biomass.

Secchi depth showed greater temporal variability, with breakpoints detected in 2007, 2010, and 2016. Mean Secchi depth increased from approximately 2.5 m (pre-2007) to 4.5 m (2007–2010), indicating improved clarity, but subsequently declined to 3.2 m from 2010-2016, and again worsened to 3.0 m from 2016 – present. The absence of a consistent directional trend suggests episodic rather than sustained shifts in water clarity.

DRP exhibited a breakpoint in 2011, after which mean concentrations declined from a mean of 7.2 µg/L to a mean of 5.8 µg/L, consistent with reduced bioavailable phosphorus in the water column in more recent years. DIN:DRP exhibited a breakpoint in 2022, after which the ratio increased from a mean of 6.0 to 16.6, indicating a shift toward greater nitrogen availability compared to phosphorus in recent years.

Overall, breakpoints were not synchronous across variables, and the timing of detected shifts differed among parameters. This lack of coherence suggests that no single event produced a consistent, system-wide structural change in water quality over the study period.

Across the key long-term monitoring variables, there were small overall improvements in water quality (e.g., decreased chl-a and nutrients) following each breakpoint, with the exception of Secchi depth which has shown an overall decrease since the 2011 breakpoint, indicating lower water clarity (Figure 3). In addition, DIN:DRP showed an increase since 2022, indicating a shift from predominantly N-limited conditions towards greater relative P limitation in recent years.

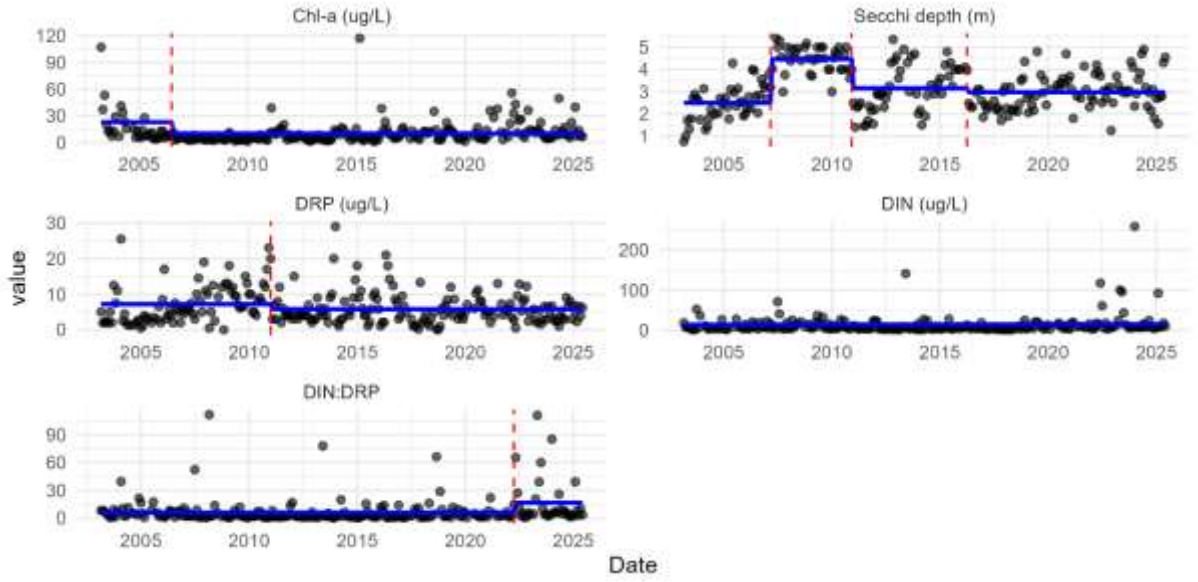


Figure 3. Time series of water quality monitoring data at Okawa Bay, Lake Rotoiti from 2003 to 2025. Vertical red dashed lines show the significant dates from breakpoint analysis. The blue lines represent the direction of the trend before and after each breakpoint (or the whole period when a breakpoint was not present). Two outliers from February 2004 (chl-a > 200 µg/L and DIN > 300 µg/L) were omitted from the figure for visual clarity but not the breakpoint analysis.

3.1.2 Physicochemical variables show that Okawa Bay is a highly dynamic, polymictic system

Monthly monitoring of water temperature and DO profiles, coupled with surface and bottom dissolved nutrient samples from Okawa Bay show that the bay is an intermittently stratified, polymictic system. Thermocline depth in Okawa Bay was highly variable over time, indicating that Okawa Bay goes from fully mixed (thermocline depth ~ 0 m) to stratified (thermocline up to 4 m depth) multiple times within a year (Figure 4a). Bottom water DO was also highly variable over time, ranging from 2.25-15 mg/L with a median concentration of 8.76 mg/L (Figure 4b): DO approached 2 mg/L on only one occasion. While low DO can enhance sediment nutrient release, the relationship is not defined by a single DO threshold and depends on sediment biogeochemistry and other environmental conditions (Waters et al., 2021). However, monthly sampling may not capture short-lived periods of low oxygen or anoxia.

The difference between surface and bottom nutrient concentrations was typically close to zero, further demonstrating that the Bay is often fully mixed (Figure 4c, d). However, some sampling events showed higher concentrations in the bottom waters, with surface-minus-bottom concentration differences reaching -50 µg/L for DRP and -186 µg/L for DIN, providing evidence that internal loading sometimes occurs. Instances of internal loading were more common for DIN than for DRP.

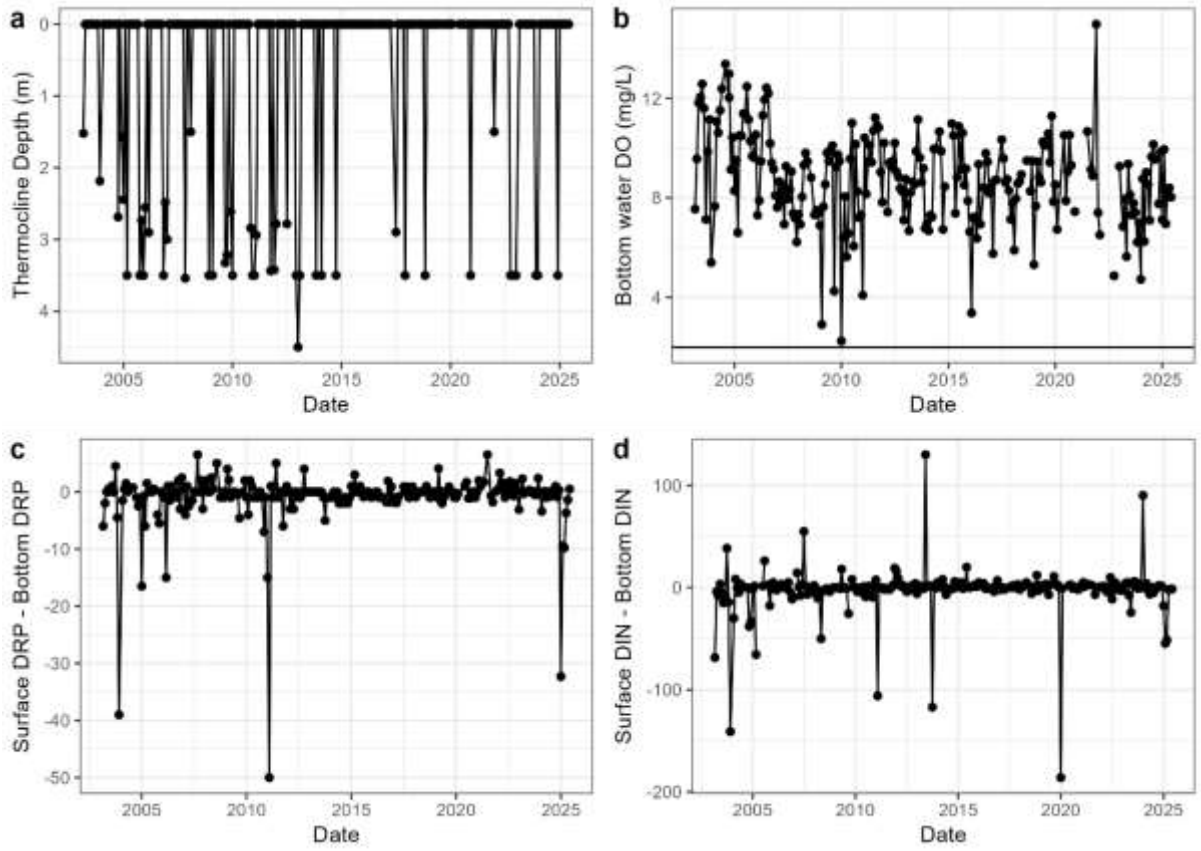


*Figure 4. Physicochemical variables from monthly monitoring over time in Okawa Bay, Lake Rotoiti with **a)** thermocline depth (minimum density gradient = 0.1 kg m^{-3} m^{-1}; mixed cutoff = 1°C) where 0 m indicates that no thermocline was detected and values >0 m represent the calculated thermocline depth, **b)** the difference between surface and bottom water DRP where positive values indicate higher surface concentrations, **c)** bottom water dissolved oxygen (DO) where the horizontal black line shows 2 mg/L, typically considered the threshold for potential internal nutrient loading, and **d)** the difference between surface and bottom water DIN, where positive values indicate higher surface concentrations.*

High-frequency (every ten minute) observations deployed in Okawa Bay over the course of approximately six weeks (18 December 2025-28 January 2026) further emphasize the polymictic nature of Okawa Bay (Figure 5). During this time period, both water temperature and DO show strong diurnal fluctuations and high variability. Several events corresponded to draw-downs in DO, with one occasion (25 December 2025) resulting in DO < 2 mg/L (min = 0.38 mg/L). Variability in DO was associated with mixing (Figure 5c), with seven mixing events (e.g., moving from a thermocline >0 to fully mixed, where the thermocline = 0) observed during this ~one month deployment.

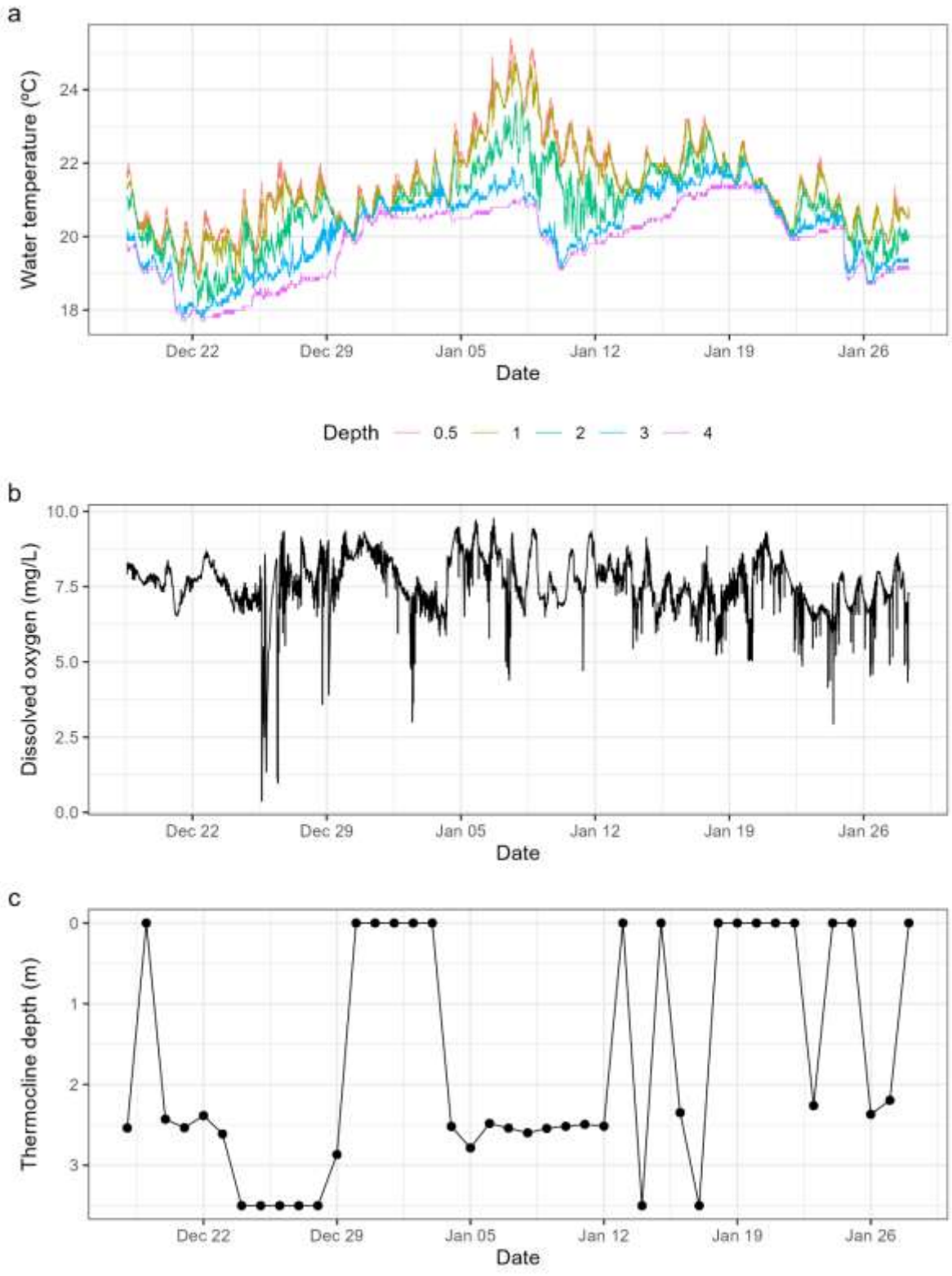


*Figure 5. High-frequency time series of **a)** water temperature at 0.5, 1.0, 2.0, 3.0 and 4.0 m, **b)** dissolved oxygen at 4.0 m, and **c)** maximum daily thermocline depth calculated from sensors deployed in Okawa Bay from 18 December 2025 to 28 January 2026.*

3.2 Comparison of water quality in Okawa Bay to other sites in Lake Rotoiti

Our results from Section 3.1 demonstrate that Okawa Bay is a polymictic system, which is sensitive to internal loading and short-term variability in physicochemical conditions. To better contextualize Okawa Bay with water quality in the much deeper, monomictic western and eastern basins of Lake Rotoiti, in Section 3.2 we compare Okawa Bay with Site 3 and Site 4.

3.2.1 Seasonal patterns in water quality varies across sites

Overall patterns observed across all data in Lake Rotoiti demonstrate clear differences in water quality across sites. Chl-a was consistently higher in Okawa Bay than in the main Lake Rotoiti basin at Site 3 or Site 4, with a long-term mean of 12.0 µg/L, compared to 7.1 µg/L at Site 3 and 5.8 µg/L at Site 4 (Figure 6; Appendix Table A2). Secchi depth was shallowest in Okawa Bay with an overall mean of 3.2 m, compared to 5 m and 6.2 m at Sites 3 and 4, respectively (Figure 6; Appendix Table A2). Surface DRP was highest and most variable at Site 4 (mean = 8.5 µg/L, sd = 8.7 µg/L), followed by Site 3 (mean = 6.4 µg/L, sd = 6.7 µg/L), and Okawa Bay having the lowest and least variable surface DRP concentrations (mean = 6.3 µg/L, sd = 4.7 µg/L) (Figure 6; Appendix Table A2).

Patterns in water quality also varied seasonally across sites. Chl-a in Okawa Bay tended to be highest and substantially more variable in autumn compared to other seasons at that site, with a mean of 21.3 µg/L (sd = 34.2 µg/L) (Appendix Table A3). The highest and most variable chl-a concentrations at Site 3 were also observed in autumn (mean = 8.8, max = 4.5 µg/L, Appendix Table A3), while winter had the highest mean (7.3 µg/L) but spring was most variable (sd = 8 µg/L) at Site 4. There was low seasonal variability in Secchi depth in Okawa Bay, with seasonal means ranging from 3.0 m in summer to 3.3 m in spring. Secchi depths across seasons were more variable at Site 3, ranging from 4.5 m in spring to 5.3 m in autumn. Site 4 was more seasonally variable yet, ranging from 5.6 m in winter to its highest mean Secchi depth of 6.8 m in summer. Surface DRP was substantially higher in winter at both Site 3 and Site 4 (mean = 14 µg/L and sd = 8.5 µg/L; 19.0 µg/L and sd = 9.3 µg/L, respectively), compared to winter mean DRP at Okawa Bay (mean = 5.7 µg/L and sd = 3.7 µg/L). Spring also showed relatively high concentrations of DRP at Sites 3 and 4, while summer and autumn had comparatively low surface water concentrations of DRP. DRP in Okawa Bay was highest during summer (mean = 8.1 µg/L and sd = 6.5 µg/L), compared to 3.3 µg/L (sd = 3.0 µg/L) and 3.4 µg/L (sd = 2.2 µg/L) at Sites 3 and 4, respectively. Patterns in surface DIN were similar to surface DRP in that concentrations were highest in winter at Site 3 and 4 (mean = 45.3 µg/L and sd = 52.6 µg/L; mean = 36.7 µg/L and sd = 58.9 µg/L, respectively; Figure 6). Mean concentrations were highest during summer at Okawa Bay (21.2 µg/L) and highly variable (sd = 64.6 µg/L), but were also elevated during winter (mean = 19.1 µg/L, sd = 27.7 µg/L).

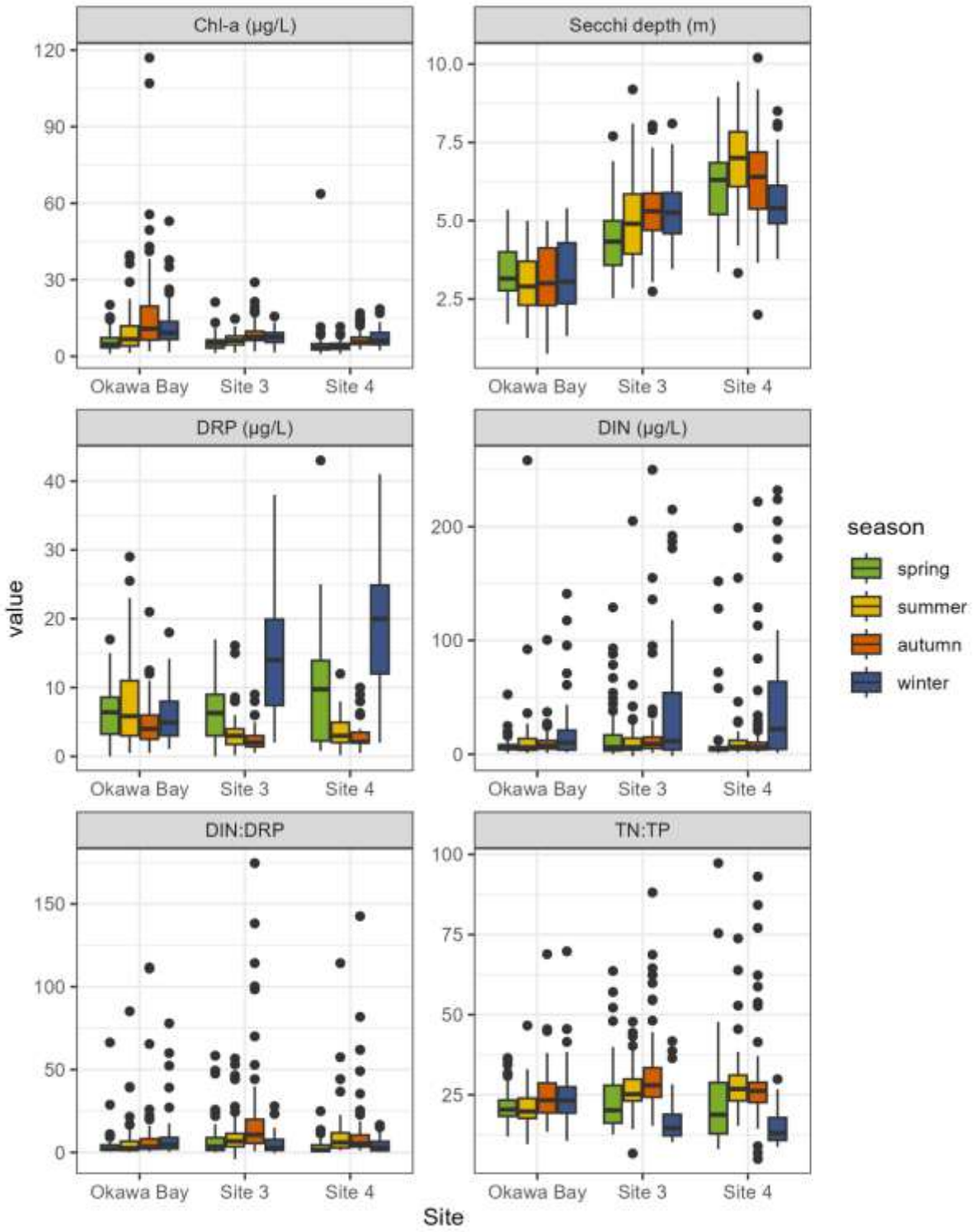


Figure 6. Boxplots of seasonal patterns in key water quality variables for Okawa Bay, Site 3, and Site 4 of Lake Rotoiti, NZ. DIN includes both NNN and NH_4. Each boxplot shows the median value as the middle line in the box, where the top of the box represents the 3rd quartile, and the bottom of the box represents the 1st quartile, the whiskers show the 1.5X the range outside the upper quartiles, and outliers greater than 1.5X outer quartiles are shown by points above the box. Two outliers from February 2004 at Okawa Bay (chl-a > 200 µg/L and DIN > 300 µg/L) were omitted from the figure for visual clarity. Note that nutrient ratios are in molar mass units for comparison with Redfield (1958) ratios.

The ratio of nitrogen to phosphorus, expressed in both dissolved (DIN:DRP) and total (TN:TP) forms, also varied across seasons and sites (Figure 6). We include both ratios to distinguish between short-term bioavailable nutrient conditions (dissolved fractions) and longer-term integrated nutrient supply (total fractions), recognising that these indicators operate at different temporal and biogeochemical scales. In this section, we interpret nutrient ratios relative to the Redfield stoichiometric benchmark of 16:1 (molar N:P) to provide a screening assessment of potential nutrient limitation (Redfield, 1958). Because Redfield ratios are expressed on a molar basis, concentrations of N and P were converted from mass units to molar concentrations prior to calculating ratios. Operationally, we interpret ratios <10 as indicative of likely nitrogen limitation, ratios between 10 and 20 as suggestive of potential co-limitation, and ratios >20 as indicative of likely phosphorus limitation. These thresholds are applied as screening indicators of potential nutrient limitation, recognising that nutrient limitation can be defined and assessed in different ways and that the selected thresholds may influence the apparent prevalence of N- versus P-limitation, rather than definitive indicators of realised phytoplankton limitation.

In Okawa Bay, DIN:DRP and TP:TP typically increased from spring through to winter, whereas DIN:DRP and TN:TP were highest during autumn at Site 3 and during summer at Site 4. Across the full-time series of nutrient ratios (Figure 7), DIN:DRP ratios indicate that Okawa Bay was potentially N-limited 82% of the time, co-limited by N and P 9% of the time and P-limited 7% of the time. Site 3 was similarly dominated by N-limitation (66%), with 18% of observations indicating co-limitation, and 13% indicating P-limitation. At Site 4, 76% of observations indicating N-limitation, 13% co-limitation, and 7% indicating P-limitation.

When looking at TN:TP, the results indicate a switch towards P-limitation, with 59% of observations demonstrating P-limitation (TN:TP >20), 40% demonstrating co-limitation, and only 0.4% indicating N-limitation TN:TP <10). Patterns were similar at Site 3, with 66% of observations indicating P-limitation, 33% indicating co-limitation, and only a single observation (0.4%) indicating N-limitation. At Site 4, the pattern was similar with 59% of observations indicating P-limitation, 32% indicating co-limitation, and 6% indicating N-limitation.

DIN:DRP is more reflective of the immediately bioavailable inorganic nutrient fractions in the water column. This ratio is sensitive to short-term processes, including recent nutrient inputs, sediment release, biological uptake, nitrification–denitrification dynamics, and mixing events. Because phytoplankton respond primarily to bioavailable nutrient pools, DIN:DRP provides a more direct indicator of potential short-term nutrient limitation. Low DIN:DRP ratios (relative to Redfield proportions) suggest nitrogen limitation or conditions favourable for nitrogen-fixing cyanobacteria, whereas high DIN:DRP ratios indicate relative phosphorus limitation under contemporary conditions.

In contrast, TN:TP represents total nutrient pools, including particulate, dissolved organic, and refractory fractions. TN:TP therefore reflects longer-term nutrient status and catchment-scale loading patterns rather than immediate bioavailable supply. Because total nutrient pools integrate historical inputs, internal recycling, and particulate storage, TN:TP is less sensitive to short-lived pulses or rapid biological uptake. Consequently, TN:TP may indicate broad trophic status or long-term nutrient imbalance even when DIN:DRP suggests short-term limitation by the opposite nutrient.



Figure 7. Time series of surface dissolved nitrogen to dissolved phosphorus (DIN:DRP) at Okawa Bay, Site 3, and Site 4 of Lake Rotoiti. The red horizontal line is at 20, above which indicates possible phosphorus limitation. The blue horizontal line is at 10, below which indicates possible nitrogen limitation. Ratios are in molar units for comparison with Redfield ratios.

3.2.2 Cyanobacterial dynamics over time across sites in Lake Rotoiti

Biovolume was higher across all sites in the 1990s, followed by decreased concentrations throughout the 2000s, and a pattern of increasing biovolume in more recent years (since 2022) at all sites. Okawa Bay has the highest overall biovolume (median = 0.11 mm^3/L), followed by Okere Arm at Kaituna (median = 0.087 mm^3/L), Te

Weta Bay (median = 0.054 mm^3/L), Hinehopu (median = 0.021 mm^3/L), and Otaramarae (median = 0.018 mm^3/L). Note that because biovolume data are highly skewed with many low and zero values, data are presented as medians, which are less sensitive to outliers than mean, and data are log-transformed for interpretation and visualisation in Figure 8.

When looking specifically at times when blooms occurred (e.g., toxic biovolume > 1.8 mm^3/L), the frequency, magnitude, and duration of blooms has varied over time and across sites (Figure 9). Blooms have occurred at all sites in Lake Rotoiti since monitoring began in 1996, but are much more common in Okere Arm and Okawa Bay (Figure 9). Specifically, since monitoring began, there have been 41 blooms observed at Okawa Bay; 35 at Okere Arm; 22 in Te Weta Bay; and only 7 at Otaramarae; and 6 at Hinehopu.

Bloom dynamics have also changed over time. Blooms occurred regularly at Oataramarae and Hinehopu in the 1990s and through 2003. Since 2003, only small blooms (max toxic biovolume < 4.2 mm^3/L), have occurred at these two sites: once in 2013 at Otaramarae and twice at Hinehopu in 2008 and 2015.

Blooms were also common in Te Weta from 1998 to 2005, with some large (max toxic biovolume observed during this time was 104 mm^3/L) and long blooms occurring, with the longest bloom lasting 108 days in 1999. Similar to Otaramarae and Hinehopu, Te Weta Bay ceased experiencing blooms from 2005 until 2014 and 2015, when three small (max biovolume < 9 mm^3/L), short (all observed only in one week) blooms occurred. Following this, there was another period without blooms, until 2022 to 2024 when 5 relatively small, short blooms were observed across these three years.

Blooms were relatively common at Okere Arm from 1998 until 2014, with one especially large (max biovolume = 135 mm^3/L), yet short (observed only in one week) bloom in 2010. No blooms were observed from 2014 until 2022 after which there was an increase in the frequency of blooms through to 2025. Since 2022, there have been 9 blooms observed in Okere Arm. Most blooms were relatively small (max biovolume < 8.3 mm^3/L), with the exception of one bloom in 2022 with a maximum biovolume of 75.7 mm^3/L which lasted for 58 days. Blooms during this latter period were typically longer in Okere Arm, with a mean duration of 17.4 days.

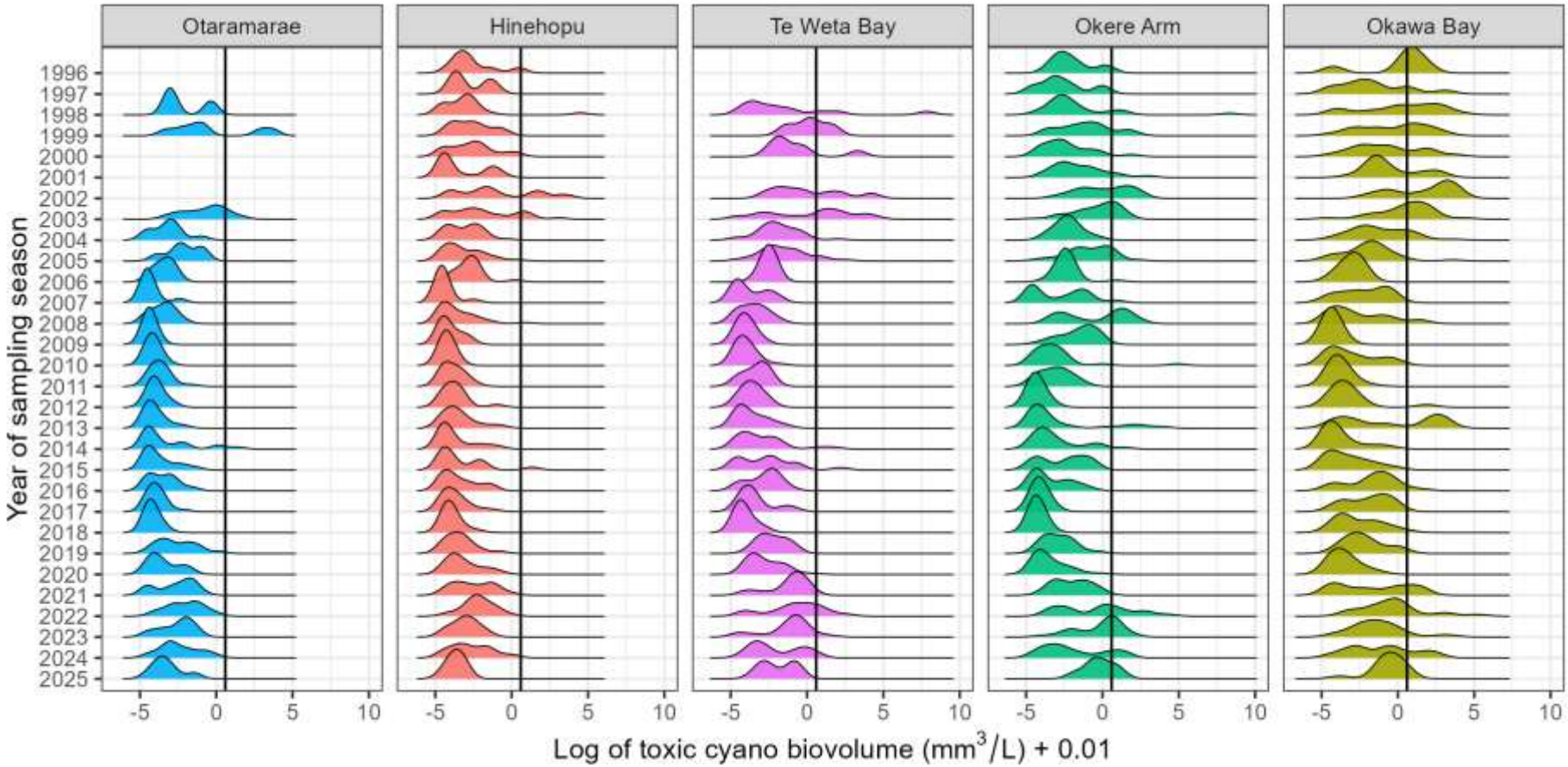


Figure 8. Density distributions of the log of toxic cyanobacteria biovolume across all five nearshore surface monitoring sites in Lake Rotoiti from 1996 to 2025. The black horizontal line represents the log of 1.8 mm^3/L threshold for issuing a safety warning based on the biovolume of potentially toxic-producing cyanobacteria. Note that 0.01 was added to all values in order to represent days with zero toxic biovolume which cannot otherwise be log-transformed. Years where distributions are not shown are years when there were fewer than two observations (e.g., Otaramarae in 2000); also note that sample size can vary between years and sites.

Okawa Bay experienced numerous blooms (n = 26) from 1996 to 2005, with at least one bloom observed every year. During this time, blooms had moderate intensity (mean of maximum biovolume = 21 mm^3/L), and length (mean duration = 22 days). Following 2005, blooms were more sporadic, with one bloom in 2008, one in 2012, two in 2013, and one in 2016 and 2018. The longest, most intense bloom during this period occurred in 2013 with a maximum biovolume of 29.4 mm^3/L and lasting for 42 days. From 2021 to 2025, at least one bloom has been observed in every year. During this period, the most extreme bloom observed occurred in 2022 with a maximum biovolume of 182 mm^3/L which lasted for 15 days. Most blooms since 2021 have been relatively short (mean duration of 11 days), with moderate biovolume (mean maximum observed biovolume of 28 mm^3/L).

Overall, these results suggest that blooms have overall decreased in frequency since early monitoring in the late 1990s across all sites in Lake Rotoiti, with a period of low bloom frequency across all sites from mid-2000s until early the early 2020s. Across Te Weta Bay, Okere Arm, and Okawa Bay, blooms show a pattern of increased occurrence but shorter duration from 2022 onward.

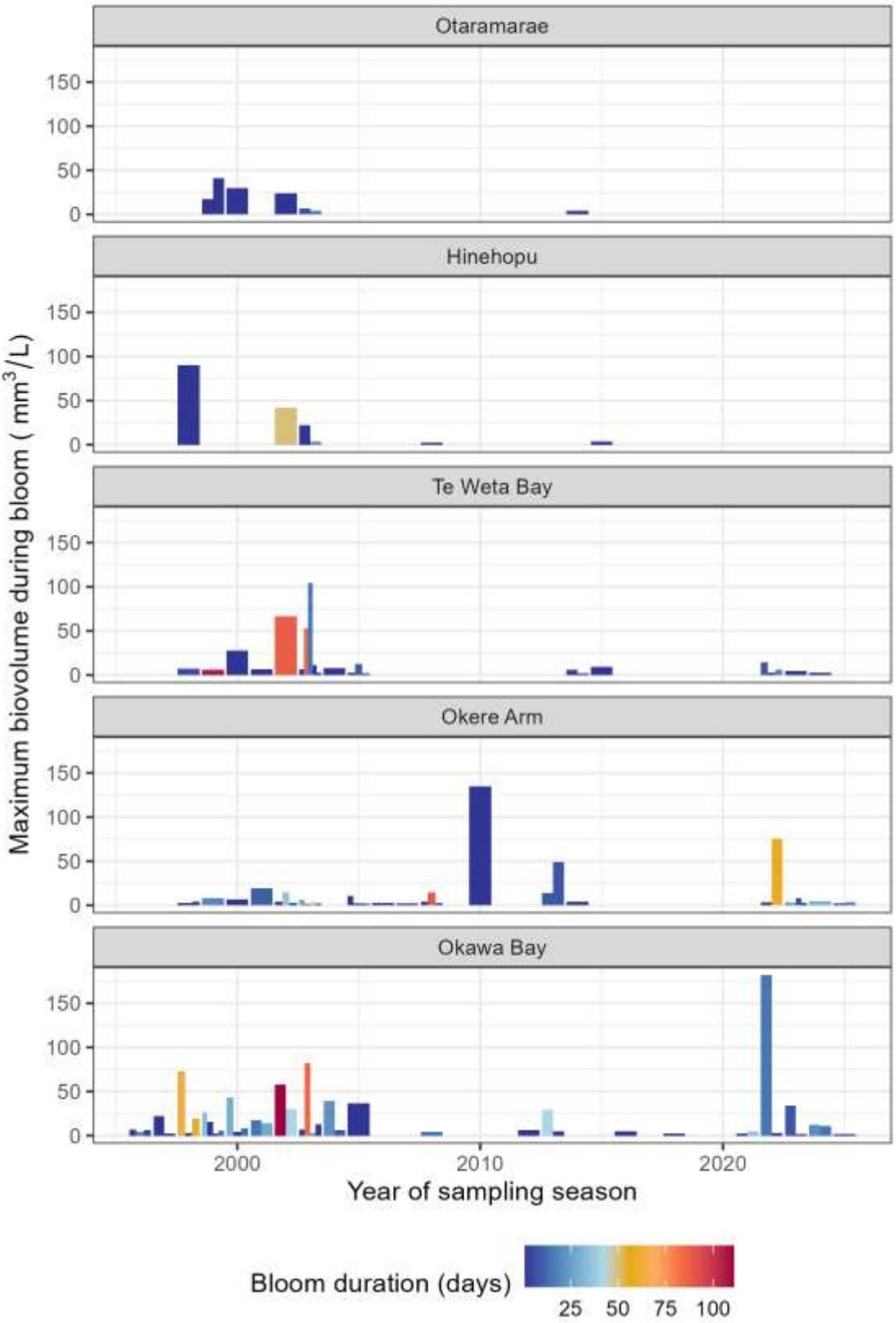


Figure 9. Frequency, duration, and magnitude of cyanobacterial blooms in Lake Rotoiti. Blooms were defined as when the amount of toxic biovolume exceeded 1.8 mm^3/L. Note that columns have variable width depending on how many blooms were observed in that sampling season.

3.2.3 Summary of cross-site comparison and implications for driver attribution

Comparative analysis across Lake Rotoiti sites shows that Okawa Bay consistently exhibits higher chl-a, higher cyanobacterial biovolume, and shallower Secchi depth (i.e., worse water clarity) than both the deeper eastern basin and other western basin sites. These differences persist across multiple years and indicate that, despite basin-scale improvements following diversion and reticulation, Okawa Bay remains disproportionately bloom-prone relative to the main basin.

Seasonal nutrient behaviour further differentiates the bay. In the deeper basin, nutrient variability is closely linked to stratification and hypolimnetic accumulation, followed by redistribution during autumn mixing. In contrast, polymictic Okawa Bay has more frequent vertical mixing and reduced separation between surface and bottom nutrient pools, resulting in less pronounced stratification-driven seasonal amplitude. This pattern reflects continuous internal redistribution of nutrients which are distributed within a shallow water column with a small volume and potential for episodic internal loading, currently not well captured by monthly sampling.

Notwithstanding these spatial differences in magnitude, cyanobacterial dynamics exhibit temporal coherence across sites. Periods of elevated bloom frequency occur in multiple parts of Lake Rotoiti during similar years, particularly in the post-2021 period. This spatial synchrony suggests that large-scale climatic and meteorological drivers influence bloom potential across the lake, while local morphometry and hydrodynamics likely determine the magnitude of response in Okawa Bay.

3.3 Understanding the main drivers of water quality in Okawa Bay

Following analyses on long-term trends in water quality at Okawa Bay (Section 3.1), and cross-site comparisons to contextualise the dynamics of Okawa Bay relative to the main basin of Lake Rotoiti (Section 3.2), in this section, we conducted a quantile regression analysis to 1) identify the most important drivers of water quality in Okawa Bay, and 2) understand if the importance of those drivers changed across extremes.

3.3.1 Quantile Regression demonstrates the importance of weather variables and within-lake nutrients on Okawa Bay water quality

Quantile regression was used to examine whether potential drivers influencing chl-a across the distribution of observed values, rather than only at the mean (i.e., ordinary least squares regression). This also provides a more nuanced examination of relationships than looking at raw data alone (e.g., Appendix Figure A1). This approach allows identification of drivers that operate preferentially under low, moderate, or high chl-a conditions. This is particularly relevant in bloom-prone systems, where extreme (upper-quantile) behaviour may be of greater management interest than average conditions.

We found significant relationships between Okawa Bay chl-a and nutrients and chl-a at Site 3, as well as several weather variables at one or more quantiles. Specifically, Site 3 chl-a showed a significant, positive association with chl-a in Okawa Bay only at 25-50% quantiles (Figure 10). DRP from Site 3 was significant at 25th quantile, while Site 3 DIN was not significant at any quantiles (Figure 10). Bottom water nutrients (DIN or DRP) in Okawa Bay were not significantly related to chl-a in Okawa Bay at any quantile, although the monthly sampling resolution may not capture short-lived internal nutrient loading events or their subsequent effects on chl-a. Median windspeed showed a significant negative relationship with chl-a at 75th-90th quantiles. Rain showed a significant positive relationship at the 50th quantile. Air temperature had no significant relationships at any quantiles, and SOI was significant at 75th-90th quantiles (Figure 10).

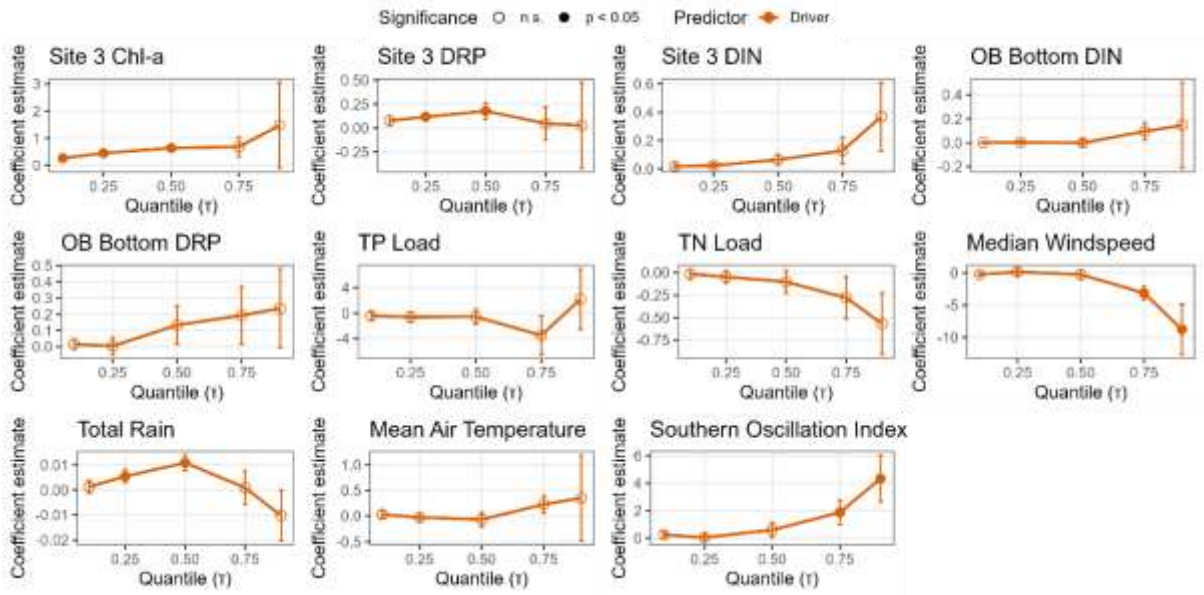


Figure 10. Quantile regression between surface chl-a in Okawa Bay and several driver variables. Significant relationships at each quantile are shown in closed circles, while insignificant relationships are shown in open circles. Significant relationships were found for chl-a and DRP from Site 3, median windspeed, total rain, and SOI. OB = Okawa Bay. TP Load and TN Load refer to total phosphorus and total nitrogen loads, respectively, from the Ōhau Channel.

After examining the time series of the significant stressors, we found that the time periods where these quantiles have occurred varies based on stressor (Figure 11). For chl-a at Site 3, the relationship with chl-a at Okawa Bay was significant throughout most of the year when chl-a concentrations at Site 3 were low (<10 µg/L), indicating that Site 3 is not acting as a source of chl-a to Okawa Bay during blooms (Figure 11). For DRP at Site 3, the relationship with Okawa Bay chl-a was strongest when Site 3 DRP concentrations were lowest, which typically occurred during the warmer months when Site 3 was stratified, from October to April (Figure 11). This further indicates that nutrients from the main basin

of Lake Rotoiti are unlikely to be contributing to higher nutrient concentrations in Okawa Bay.

Chl-a in Okawa Bay was most strongly affected by median windspeeds and SOI at the upper quantiles (75th-90th) of these variables. Windspeed showed a significant negative relationship at these quantiles (Figure 10), indicating that high wind conditions are associated with reduced probability of extreme surface chl-a concentrations in Okawa Bay. This is likely reflective of the influence of windy conditions in enhancing vertical mixing, which may redistribute phytoplankton through a greater proportion of the water column and reduce surface accumulations. Wind-driven circulation may also increase water exchange with the western basin and shorten residence times in Okawa Bay. In contrast, the relationship between SOI and chl-a was positive. Positive SOI values correspond to warmer, wetter conditions in the region, suggesting that broad-scale climatic variability may create favourable conditions for bloom development in Okawa Bay.

Rain had strongest influence on chl-a in Okawa Bay at the median (50th) quantile rather than at upper or lower tails. This suggests that moderate rainfall conditions are associated with variability in typical chl-a levels, whereas extreme rainfall events (e.g., flood-scale inputs) do not show a detectable relationship with high chl-a levels at the monthly resolution of this dataset.

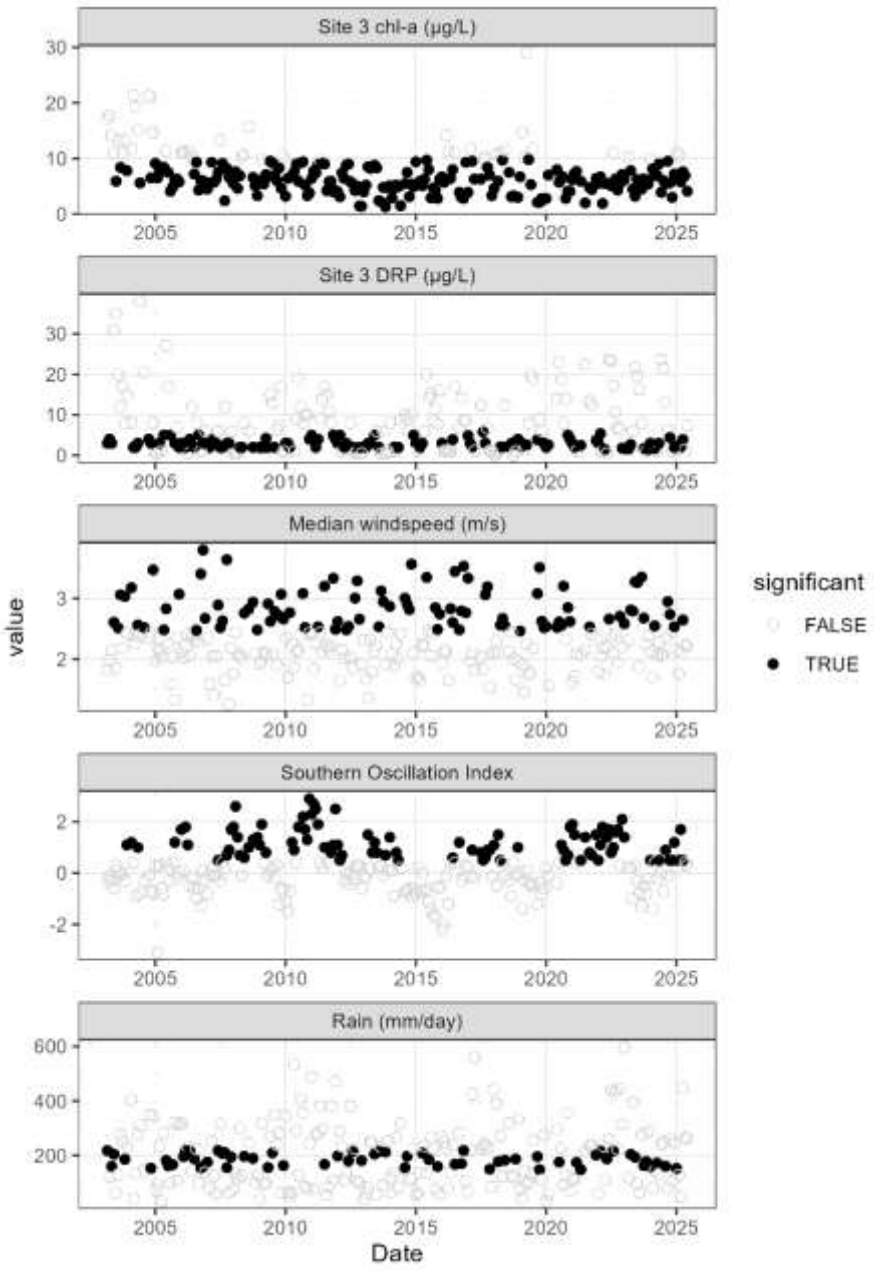


Figure 11. Time series of significant drivers from quantile regression analysis. Data are coloured in black when their magnitude corresponds to a significant quantile (based on results in Figure 10).

3.3.2 Wind patterns govern dynamics between Okawa Bay and Lake Rotoiti main basin

To better understand how physical connectivity between Okawa Bay and Site 3 may be moderated by meteorological conditions, we examined the relationship in chl-a concentrations between the two sites in relation to prevailing wind direction and wind speed. We focused on NE and SW winds because these are most directly aligned with the longitudinal axis connecting Site 3 and Okawa Bay. Northeasterly (NE) winds could promote surface transport from Site 3 into Okawa Bay, whereas southwesterly (SW) winds would favour transport out of Okawa Bay towards Site 3. This does not preclude exchange under other wind directions, which may generate more complex circulation

within and around the bay. We examined the relationships between chl-a at these two sites based on the prevailing wind direction from the previous 3 days before monthly nutrient sampling occurred.

The association between the two sites was stronger when winds were aligned along the NE–SW axis (Figure 12), consistent with enhanced surface transport during these periods. In contrast, when winds were oriented perpendicular to this axis (NW-SE), the relationship generally weakened, suggesting reduced advective coupling. However, observations under NW-SE winds were comparatively rare (15%) relative to NE-SE aligned winds (Figure 12), limiting our ability to draw inference about what happens when the winds are not favourably aligned.

Most monthly sampling events occurred during periods characterised by relatively low wind speeds, with strong wind events being comparatively infrequent (Figure 12). Wind magnitude did not exhibit a clear secondary effect on the strength of the inter-site relationship, indicating that wind direction is more important than absolute wind speed for facilitating exchange between basins.

Importantly, under NE wind conditions, when transport into Okawa Bay would be expected, chl-a concentrations at Site 3 exceeded those in Okawa Bay only when concentrations were relatively low (approximately <9 μg L^{-1}). At higher chl-a, Okawa Bay typically exhibited greater concentrations than Site 3, even during periods of potential inflow. This pattern suggests that advective input from Site 3 does not explicitly drive elevated bloom conditions in Okawa Bay. Under SW wind conditions, when surface transport from Okawa Bay toward Site 3 would be expected, no consistent or systematic pattern of directional “seeding” from the bay into the main basin was observed, except for some discrete periods (Figure 12), when chl-a concentrations in Okawa Bay were up to approximately two-fold higher than those at Site 3.

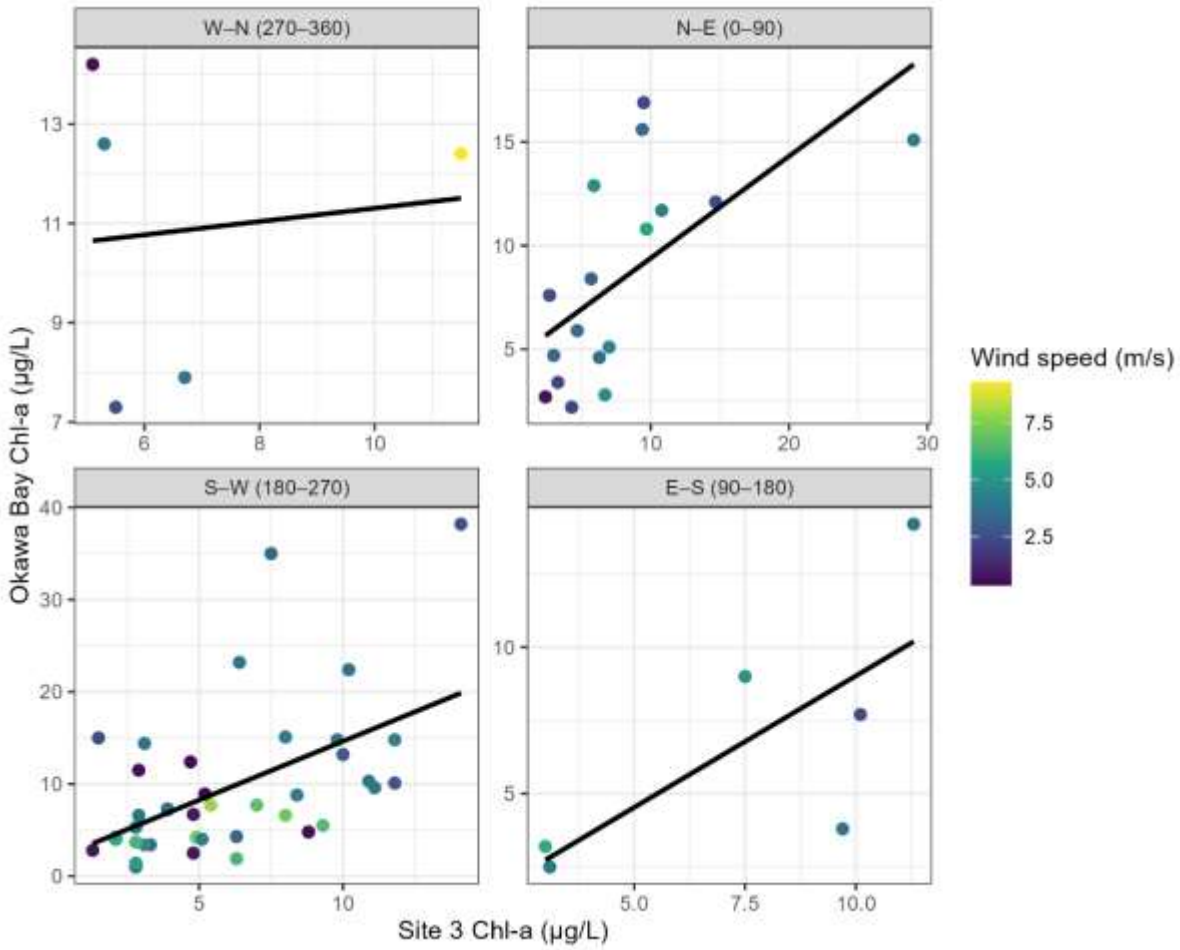


Figure 12. Patterns between chl-a at Site 3 and Okawa Bay, broken down by wind direction and colored by wind speed. Wind dynamics are calculated as the average wind patterns observed in the 3 days preceding the nutrient observations in Okawa Bay. One outlier in Okawa Bay chl-a of 200 µg/L has been removed for visualisation purposes.

3.4 Examining changing relationships with Okawa Bay water quality over time

In Section 3.3, we identified potential key drivers which regulate water quality in Okawa Bay. To extend this analysis, Section 3.4.1 will directly examine how changes in the Ōhau Channel diversion wall over time have altered relationships between water quality (TN, TP, and chl-a) in Okawa Bay and key sources of nutrients (external nutrient loads from Ōhau Channel, internal nutrients from Site 3). Section 3.4.2 extends this further by using a moving window analysis to identify how the importance of driver variables changes over 8-year time periods following methods in Woelmer et al. (2025).

3.4.1 Changes in relationships between external and internal nutrients in response to Ōhau Channel diversion wall

Prior to installation of the Ōhau Channel diversion wall in 2008, we hypothesise that water quality in Okawa Bay was positively related to both external loads from the Ōhau Channel (H_{ext}) and nutrient concentrations in the western basin at Site 3 ($H_{Site\ 3}$; Figure 13a).

Following the installation in 2008, we hypothesise that the relationship between external loads and Okawa Bay water quality was reduced to near zero, while the relationship between internal loads and Okawa Bay may have strengthened due to increased internal residence time and nutrient cycling (Figure 13b).

From 2020 onwards, when holes in the diversion wall were confirmed to be substantial, we hypothesize that there would again be a positive relationship between external loads and Okawa Bay water quality, but likely lower than the pre-2008 relationship. Similarly, we hypothesise that the positive relationship between internal nutrients and Okawa Bay water quality would remain positive, but decreased from the 2008-2020 time period, and increased from the pre-2008 time period (Figure 13c).

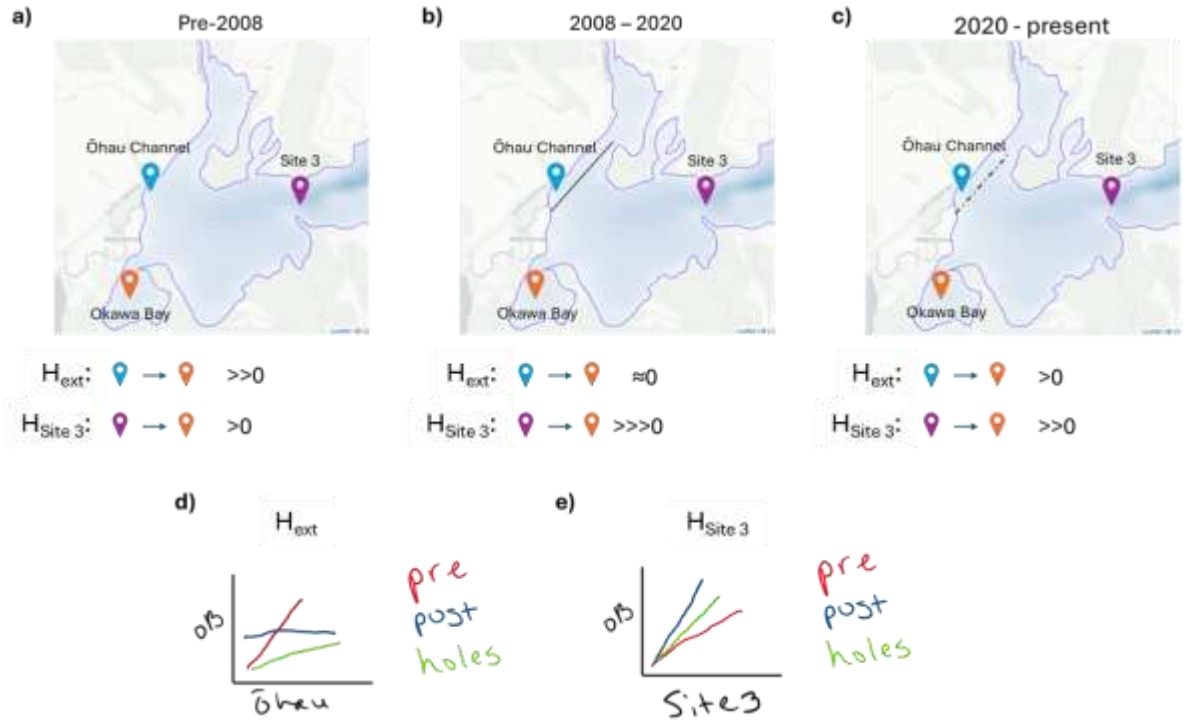


Figure 13. Conceptual representation of the hypothesised changes in relationships between external (Ōhau Channel) and internal (Site 3) nutrient and chl-a sources to Okawa Bay (OB).

To test these hypotheses, we qualitatively compared the relationships of TN, TP, and chl-a between Okawa Bay and Ōhau Channel (representing external nutrient loads) and Site 3 and visually assessed the slope and intercept of a linear model fit over each time period.

For external loads, pre-wall relationships between Okawa Bay chl-a and Ōhau Channel TN and TP were slightly negative (Figure 14), contrary to our hypothesis that were would be a positive relationship before wall installation (Figure 13a, d). However, we found that external TN and TP generally exhibited a near zero relationship during the post-wall time period (Figure 14), and this relationship became positive following the confirmation of holes in 2020 (Figure 13b, c, d). Chl-a data from Ōhau Channel was not available prior to 2011, but we found that the relationship between external chl-a and Okawa Bay remained consistent between post-wall and post-hole time periods, with a slightly higher intercept post-hole period (Figure 14), indicating overall higher concentrations of chl-a during this period.

For nutrients at Site 3, we found a positive relationship between Site 3 TN and Okawa Bay chl-a (Figure 14) consistent with our hypothesis (Figure 13 a, e), but not for Site 3 TP during the pre-wall time period. Generally, across all time periods, there was a near-zero relationship between TP at Site 3 and Okawa Bay chl-a (Figure 14). Post-wall, the relationships of Site 3 TN and Site 3 chl-a with Okawa Bay chl-a decreased, to near-zero for TN (Figure 14), contrary to our hypothesis (Figure 13b, e). Post-holes, the slope and intercept for all relationships increased slightly (Figure 14), consistent with our hypothesis (Figure 13c, e).

Generally, there appears to be an impact of the wall on the relationship with both Ōhau Channel nutrient loads and Site 3 nutrient concentrations for Okawa Bay chl-a, the strength of which varies by variable. Some patterns were contrary to expectations, for example, negative relationships with external nutrient loads and Okawa Bay prior to wall installation. In addition, we did not expect to see a decrease in the post-wall importance of nutrients and chl-a from Site 3. Overall, the relationships generally appeared stronger post-2020 than during the preceding post-wall period. While this is consistent with possible changes in connectivity between Okawa Bay and the western basin, other factors contributing to these temporal differences cannot be excluded.

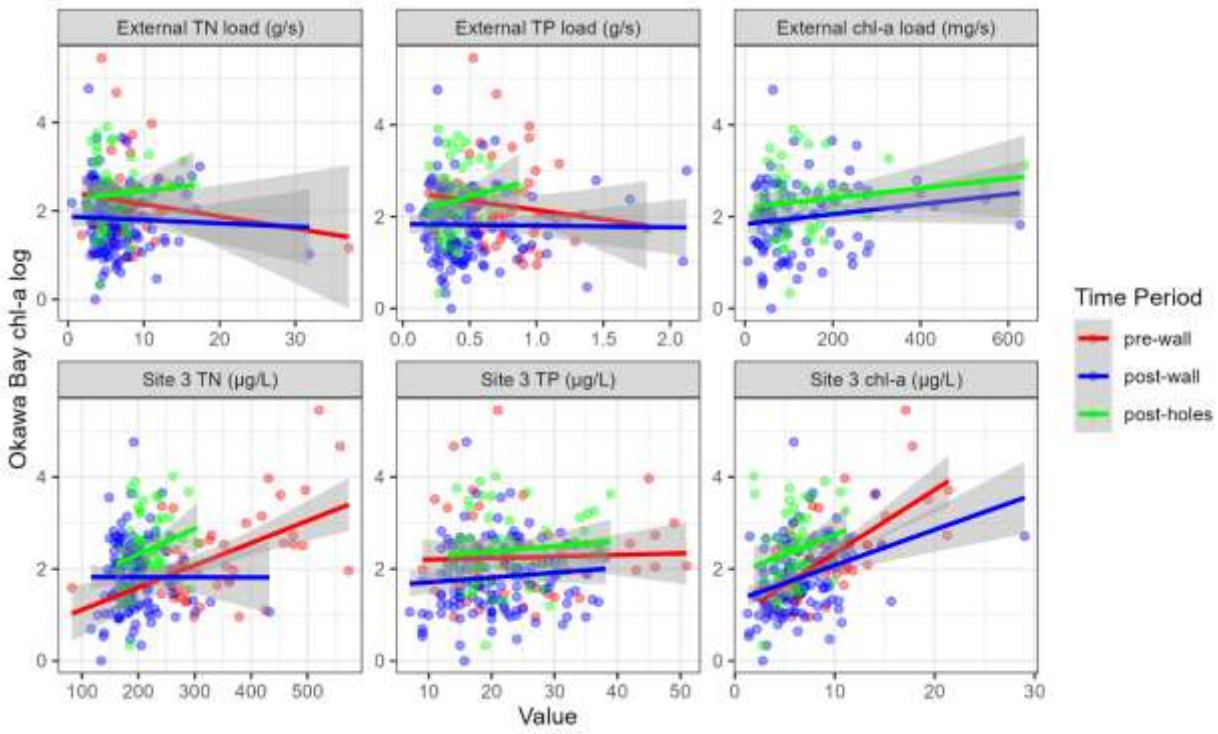


Figure 14. Relationships between Ōhau Channel TN, TP, and chl-a loads, TN, TP, and chl-a concentrations at Site 3a in relation to Okawa Bay chl-a. Relationships are broken down by pre-wall (pre 2008-01-01), post-wall (2008-01-01 to 2020-01-01) and post-holes (2020-01-01 to present). Note that chl-a data from the Ōhau Channel are not available prior to 2011. Ōhau Channel loads were calculated by multiplying concentrations measured by the corresponding daily mean discharge (TN and TP: g/s; chl-a: mg/s).

3.4.2 Moving window analysis shows that drivers of water quality vary over time in Okawa Bay

Moving window analysis on Okawa Bay chl-a demonstrates that the importance of different variables changes over time in the bay (Figure 15), but that overall explanatory power of the individual models was relatively low (mean R^2 = 0.2). The dates shown in Figure 15 represent the start of each 100-month (~8-year) moving window; consequently, the final windows beginning around 2015 include observations extending into the 2020s.

Based on the relative rankings of each driver variable, we found that bottom water DRP has increased in importance in the last ~10 years of moving windows. This coincides with an increased importance of thermocline depth and air temperature. Over the same period (Figure 4), the frequency and persistence of stratification appear to have decreased. This suggests that vertical connectivity between bottom and surface waters may have increased, allowing bottom-water DRP to exert a stronger influence on surface nutrient availability and, consequently, chl-a dynamics. In this context, bottom-water DRP may serve as a proxy for episodic internal phosphorus supply to the photic zone.

In contrast, bottom-water DIN has declined in relative importance, while phosphorus-related variables have increased. This shift is consistent with changes in DIN:DRP ratios, which indicate an increasing tendency toward phosphorus limitation and a reduced likelihood of nitrogen limitation in recent years. Supporting this interpretation, Wood et al. (2022) report a decline in the proportion of N-fixing cyanobacteria in Lake Rotoiti, which may reflect reduced ecological advantage for N-fixers under conditions of relatively higher nitrogen availability. However, it is important to note that Wood et al. (2022) analysed the main basin of Lake Rotoiti, and their findings may not directly represent conditions within Okawa Bay.

Meteorological drivers also exhibit temporal shifts in relative importance. Windspeed and SOI show intermittent periods of increased and decreased influence. Periods when SOI is ranked as important correspond to phases of elevated SOI (Figure 11), suggesting that large-scale climate variability may modulate chl-a responses during certain intervals. This interpretation aligns with the quantile regression results, which indicate that extreme SOI conditions are particularly influential for chl-a dynamics. During periods when SOI is of lower relative importance, windspeed similarly declines in influence, consistent with reduced wind-driven mixing under less extreme climate conditions. Negative SOI values, often associated with windier conditions, further support the importance of wind-related mixing processes in structuring chl-a variability (Figure 10). The declining importance of rainfall over time suggests that external nutrient inputs via precipitation-driven runoff may have become a less dominant control on surface chl-a relative to internal loading and physical mixing processes. This pattern may reflect changes in catchment inputs, hydrological connectivity, or the increasing relative importance of internal nutrient cycling within the bay.

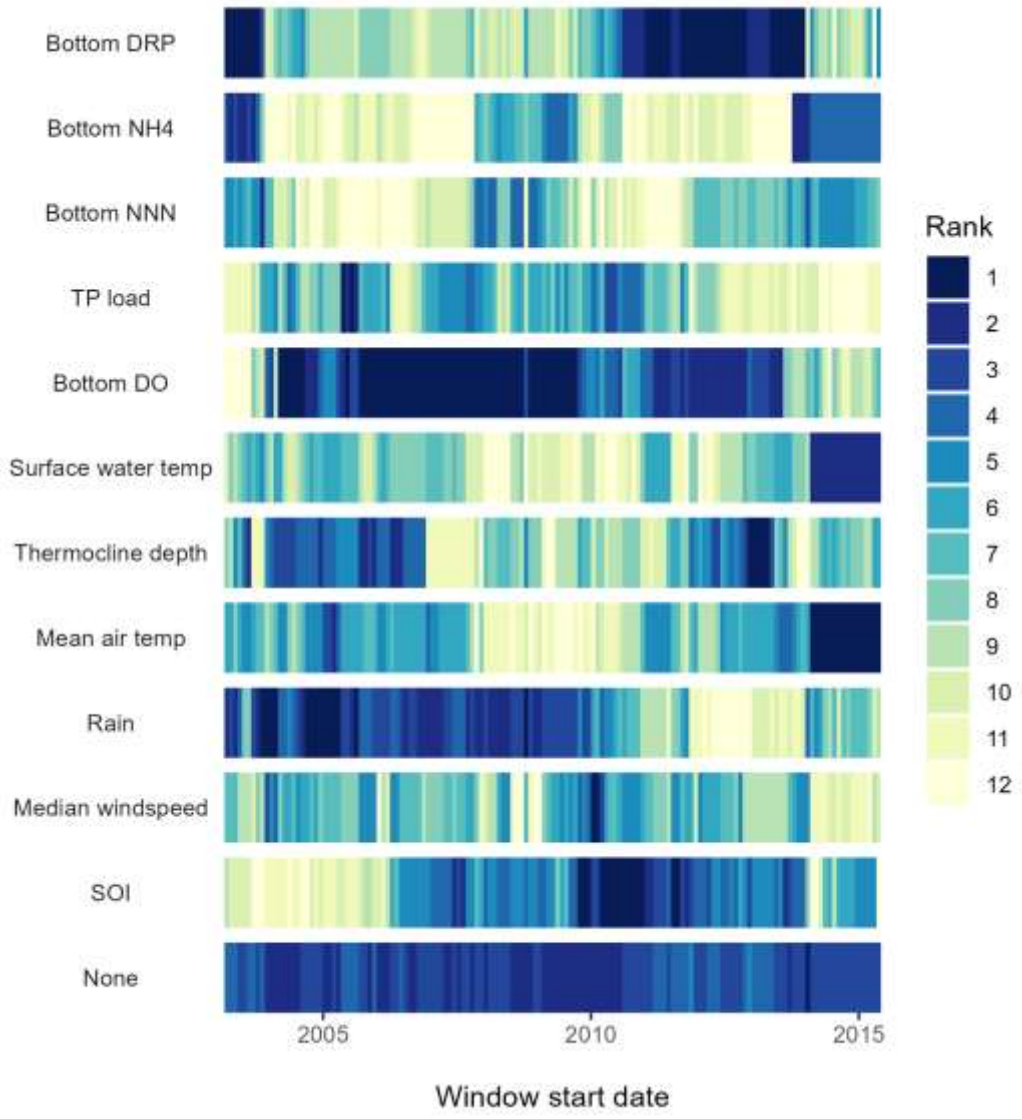


Figure 15. Driver importance rankings from a moving window analysis, where 'none' represents the ranking of a model with only autoregressive components (i.e., just chl-a from the previous month) as a control. Darker colours indicate greater importance of the driver variable. Dates indicate the start of each 100-month moving window.

3.4.3 Assessment of weed spray effects on nutrient dynamics in Okawa Bay

To assess whether aquatic weed spraying was associated with detectable changes in whole-bay nutrient mass, we compared post-event rates of change in TP and TN between spray and non-spray years. Rates of change were expressed in kilograms per day over the 120-day period following each event, relative to a 60-day pre-event baseline. The 120-day window was selected to encompass the assumed period of macrophyte decomposition. However, given the much shorter water-exchange timescales in Okawa Bay and the monthly sampling frequency, this analysis was intended to identify sustained changes in water-column nutrient mass rather than quantify the actual nutrient release from decaying macrophytes.

Whole-bay TP and TN mass exhibited substantial interannual variability across the monitoring record. When aligned to spray dates, post-event trajectories did not show a consistent pattern of sustained nutrient accumulation relative to non-spray years.

The macrophyte decay model generated predicted cumulative nutrient release within the uncertainty bounds defined by Scholes (2015a). However, predicted cumulative nutrient release frequently exceeded the observed net changes in whole-bay nutrient mass for both TN and TP (Figure 16). In several spray years, predicted release implied positive slopes over the 120-day window, while observed slopes were near zero or negative. Direct comparison between observed and modelled slopes demonstrated poor alignment across years. The differences varied in magnitude and direction among spray events (Appendix Figure A2). There was no consistent pattern of agreement across spray events. Importantly, modelled cumulative release and observed net changes in nutrient mass are not directly equivalent, as the latter also reflect nutrient uptake, sedimentation, transport and other inputs and losses.

Comparison of post-event slopes between spray and non-spray years showed no statistically distinguishable difference for either TP or TN. Confidence intervals for the difference in slopes overlapped zero in both cases, indicating that post-spray rates of change were not systematically greater than seasonal background variability (Appendix Figure A3). Overall, the magnitude of interannual variability in TP and TN mass during non-spray years was comparable to that observed during spray years.

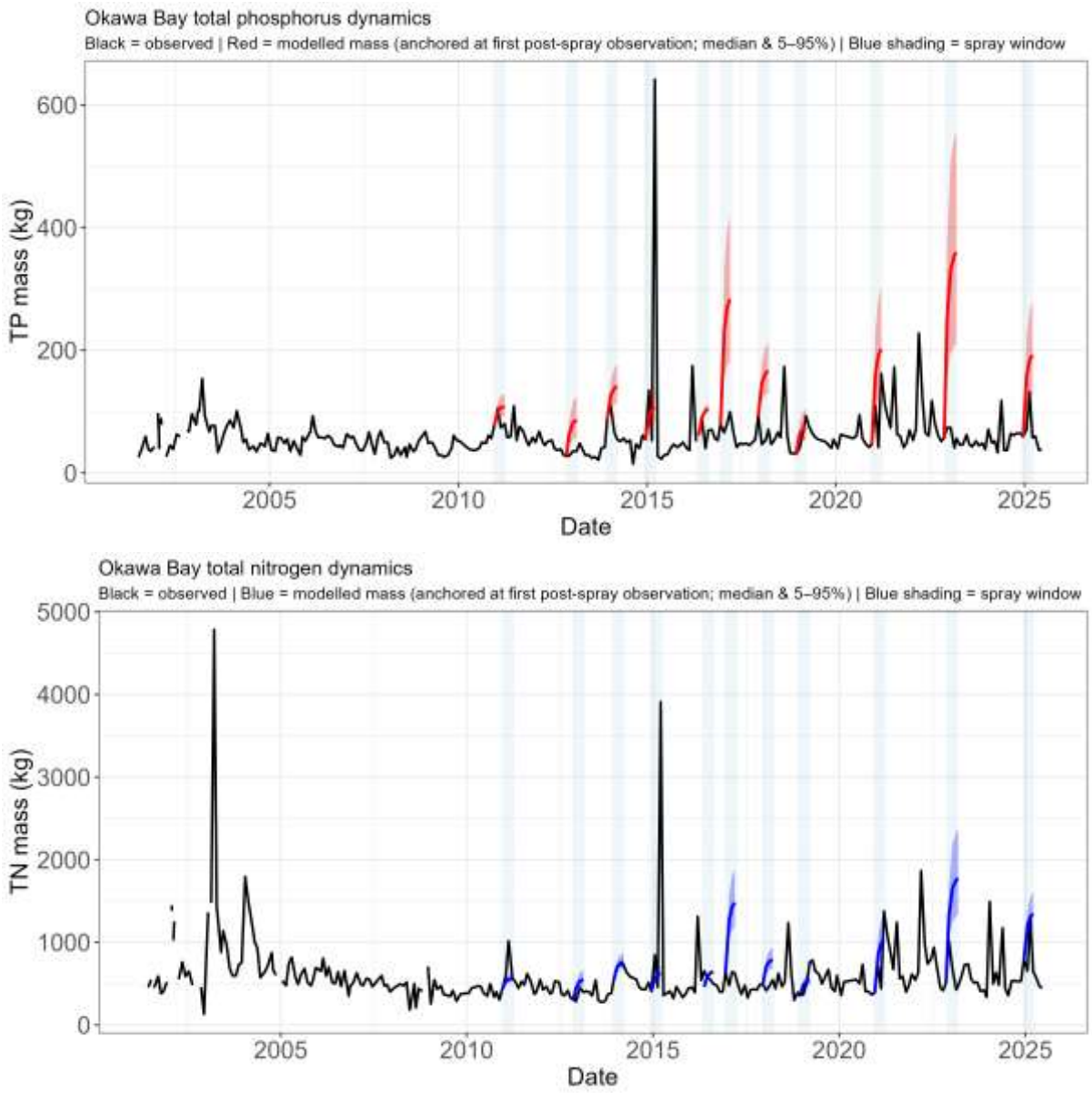


Figure 16. Time series of total phosphorus (TP; top) and total nitrogen (TN; bottom) mass (kg) in Okawa Bay. Black lines represent observed nutrient mass. Red and blue lines indicate modelled median internal release, with shaded bands showing the 5th–95th percentile prediction interval. Light blue shading denotes 120-day post-spray windows. Modelling starts in 2010, when the first spray data was available at the time of writing.

The analysis indicates that predicted nutrient release from macrophyte decay does not consistently match the magnitude or direction of observed post-spray changes. In some years the model overestimates the observed rate, in others it underestimates it, although the agreement was generally closer for TN than TP. This suggests that while decay may contribute nutrients locally, its signal is not clearly detectable above system-wide variability at the temporal resolution of monthly monitoring. We considered whole-bay TN and TP mass to assess whether potential nutrient release from decaying macrophytes was associated with detectable changes in the bay's nutrient pool, rather than as a direct

measure of the amount released. Dissolved inorganic nitrogen and DRP may respond more directly to nutrient release during macrophyte decomposition, as considered by Scholes (2015b), but the available monitoring was considered not well suited to testing this response. The decay model predicts that approximately half of the nutrient pool is released within the first 20 days and ~66% within 30 days, whereas most water-quality samples are collected monthly. Any DIN or DRP released during this period may also be rapidly taken up, transformed, mixed through the bay or transported out of Okawa Bay. A monthly DIN or DRP concentration therefore may not provide a good comparison with the cumulative nutrient release predicted by the decay model. Resolving this response would likely require more frequent sampling during the first weeks following spraying.

4. Discussion

The purpose of this investigation was to provide BOPRC with a data-driven assessment of the drivers influencing phytoplankton biomass and cyanobacterial bloom behaviour in Okawa Bay, within the broader context of Lake Rotoiti and management actions taken by BOPRC. We demonstrate that the shallow, polymictic nature of Okawa Bay makes it a dynamic system which is sensitive to changes in meteorology, climatic variability, and changes in hydrodynamics with the main basin. Because nutrients remain elevated throughout most of the year, Okawa Bay is susceptible to algal blooms during periods when environmental conditions align.

4.1 Primary drivers of chlorophyll-a in Okawa Bay

Quantile regression analysis demonstrates that chl-a in Okawa Bay is strongly associated with meteorological forcing, particularly windspeed, and with broader climatic variability as represented by SOI. Importantly, these relationships are quantile-specific, where windspeed is a statistically significant negative predictor at upper quantiles of chl-a ($\tau \geq 0.75$), indicating that extreme bloom states are constrained by windy, mixed conditions. Because this relationship emerges specifically at higher quantiles, it supports the interpretation that wind acts primarily as a regulator of bloom intensity rather than background productivity. This is consistent with shallow lake limnology, where increased wind-driven turbulence reduces stratification stability, and suppresses sustained phytoplankton accumulation (Reynolds, 2006; Paerl & Huisman, 2009). Note that in shallow systems dominated by wind-driven sediment resuspension (e.g., exposed Danish lakes; Søndergaard et al., 2003), wind speed often shows a positive relationship with turbidity and nutrient release once shear thresholds are exceeded. However, in Okawa Bay, we observed a negative relationship between windspeed and chl-a. The relatively limited fetch of Okawa Bay, particularly in relation to its depth, may limit the extent and persistence of wind-driven sediment resuspension compared with more exposed shallow lakes. This is consistent with resuspension not being the dominant

mechanism controlling bloom magnitude, although sediment resuspension was not explicitly examined in our analysis.

The significant association between SOI and upper-quantile chl-a suggests that large-scale climatic variability may contribute to conditions associated with high chl-a in Okawa Bay. However, the relatively low explanatory power of the quantile regression models indicates that SOI accounts for only a small component of the variability in chl-a, and the relationship should therefore not be interpreted as a strong predictor of bloom severity. Comparable patterns have been reported globally, where warming and altered stratification regimes can favour cyanobacterial blooms even under stable nutrient loads (O'Neil et al., 2012; Woolway et al., 2021). Atmospheric stilling may provide an additional mechanism, with reduced wind-driven turbulence favouring the formation and surface accumulation of buoyant cyanobacteria (Ranjbar et al., 2022). In Okawa Bay, the temporal coincidence of post-2021 cyanobacterial bloom resurgence with climatic variability is consistent with a possible climatic influence but does not establish a causal relationship. Future analyses could examine bloom metrics by SOI phase and test whether upper-percentile chl-a or bloom frequency differs systematically between climatic states.

Chl-a and DRP from Site 3 were significant predictors of chl-a in Okawa Bay at low quantiles, indicating that there is a relationship between the dynamics happening at these two sites only under non-bloom conditions. While our wind analysis demonstrates that water exchange between the two sites may be enhanced under favourable wind directions, we cannot exclusively prove that exchange is occurring under these conditions without high-frequency monitoring of water flow and velocity at the inlet to Okawa Bay. Given that chl-a at Site 3 tends to be lower than chl-a in Okawa Bay, we hypothesize that this relationship is driven by local seasonal patterns which simultaneously impact both sites within Lake Rotoiti. As a result, this association is likely driven by temporal synchrony and may be facilitated by favourable wind conditions.

Bottom-water nutrients within Okawa Bay were not statistically significant predictors of chl-a at monthly resolution. This result does not imply that internal loading is absent in the bay, but that nutrient-biomass coupling occurs at shorter temporal scales, which are not captured by monthly sampling in polymictic Okawa Bay. Evidence from nearby Lake Rotorua demonstrates that intermittent stratification events (defined as a temperature difference ≥ 0.25°C between surface (0.5m) and near-bottom (20.5m) waters, lasting ≥3 days) occurred up to 18 times per hydrological year, with sustained events occasionally lasting up to 58 consecutive days (McBride 2022). Stratification events such as these regularly led to bottom-water anoxia at least once in most years, causing increased bottom-water DRP and ammoniacal nitrogen after approximately 7-10 days of continuous stratification. Importantly, annual mean surface TP in Lake Rotorua was significantly related to stratification structure ($R^2 = 0.403$ for short-term events; $p < 0.02$), demonstrating that stratification metrics in such a polymictic lake can explain a

substantial proportion of interannual phosphorus variability even in a larger and deeper basin. Given that Okawa Bay is shallower (~4 m) and more sheltered, the time required to induce hypoxia following stratification may be shorter than in Lake Rotorua, potentially increasing the sensitivity of the bay to calm, stratified periods. The absence of strong monthly-scale coupling between bottom nutrients and chl-a in Okawa Bay is therefore not inconsistent with active internal loading. Rather, it reflects the likelihood that nutrient release events are episodic and rapidly redistributed following mixing, such that event-scale fluxes are not captured by routine monthly sampling.

Changes in nutrient stoichiometry provide additional context for interpreting chl-a dynamics in Okawa Bay. The recent increase in DIN:DRP indicates that DIN has become relatively more available compared to DRP. Elevated DIN:DRP is commonly interpreted as increasing relative phosphorus limitation in freshwater systems (Schindler, 1977; Søndergaard et al., 2003), although these changes can arise through altered sediment–water phosphorus exchange, denitrification, or differential biological uptake (Nürnberg, 2009).

Ratios of TN:TP remain persistently high in Okawa Bay, indicating that TN is abundant relative to TP. While dissolved ratios reflect short-term bioavailable pools and total ratios integrate longer-term loading, storage, and recycling processes (Abell et al., 2015), the two ratios suggest different patterns of nutrient availability, with DIN:DRP indicating potential nitrogen limitation across much of the historical record and TN:TP indicating relatively greater nitrogen abundance in the total nutrient pool. This suggests that nitrogen supply is unlikely to be the dominant regulator of bloom magnitude under recent conditions. Interpreting these ratios alongside phytoplankton composition further clarifies this pattern. Although Wood et al. (2022) reported a decline in the relative abundance of N-fixing cyanobacteria across sites in Lake Rotoiti up to 2022, species-level data from Okawa Bay show that peak bloom events remain strongly dominated by *Dolichospermum* taxa. The presence of N-fixing taxa indicates that the community retains the functional capacity for nitrogen fixation. However, fixation is energetically costly and regulated by ambient dissolved nitrogen concentrations (Grover et al., 2022). These findings suggest that dissolved nitrogen is unlikely to be strongly limiting during these periods. The results indicate that bloom intensity is plausibly driven by phosphorus accessibility interacting with favourable thermal stability and reduced mixing, rather than by a fundamental shift in the limiting nutrient.

4.2 Frequency, magnitude, and temporal patterns of cyanobacterial blooms

Bloom records in Okawa Bay demonstrate a clear temporal structure: elevated bloom frequency in the late 1990s and mid-2000s, a marked decline between approximately 2006 and 2020, and renewed increases after 2021, including comparably high events in

2022. This pattern is supported by breakpoint analyses showing decreases in chl-a in the mid-2000s (Figure 3).

The decline in bloom frequency in the mid-2000s is consistent with reduced local external nutrient loading via sewage reticulation in Okawa Bay at this time (Gibbs et al. 2003) and negligible external nutrient inputs from the Ōhau Channel after completion of the diversion wall. This is also consistent with lake responses following wastewater diversion elsewhere (Søndergaard et al., 2003; Jeppesen et al., 2005), including Lake Rotorua (Rutherford et al. 1989; Abell et al. 2015; McBride et al., 2019). Although no single sustained regime shift was detected coincident with a specific intervention in Okawa Bay (see Figure 3), the reduction in bloom frequency and upper-percentile cyanobacterial biovolume during the mid-2000s to late-2010s period suggests a real, if gradual, moderation of bloom severity. However, the renewed increase in bloom frequency after 2021 occurred without clear increases in long-term nutrient concentration trends.

It is also important to consider whether structural changes to the Ōhau Channel diversion wall may be contributing to contemporary dynamics. There is evidence that the holes in the wall have enlarged over time, with measurable impacts of wall leakage at Site 3. In principle, increased leakage could enhance hydrodynamic exchange between Ōhau Channel nutrient loads and Okawa Bay. Indeed, we did observe changes in the relationships between external and internal nutrients in Okawa Bay across time periods related to the wall's progression (Figure 14). However, the available evidence does not allow us to determine the extent to which increased leakage has contributed to the recent bloom resurgence. First, no corresponding change or sustained increase in nutrient concentrations was detected in Okawa Bay coincident with the period of increased leakage. Second, tracer analyses in previous studies indicate that while hydrological coupling remains measurable, the magnitude of exchange under post-diversion conditions is substantially lower than pre-diversion inflow scenarios (Muraoka et al. 2010; von Westernhagen 2010). However, given that predominant wind directions often align to facilitate connections between Okawa Bay and the western basin, further monitoring to capture exchange dynamics is necessary to draw robust conclusions. Third, phytoplankton community composition does not support a strong seeding effect from upstream systems. Species assemblages associated with Lake Rotorua remain compositionally distinct from those dominating Okawa Bay blooms in recent years (James Dare, unpublished information). This weak taxonomic coherence argues against direct advective seeding as a dominant mechanism, although it does not exclude the transport of nutrients through wall leakage.

This decoupling between nutrient trajectories, cyanobacteria taxonomy, and bloom severity is consistent with observations overseas where climatic variability intensifies bloom expression even when nutrient loads remain stable or decline (Paerl & Huisman, 2009; O'Neil et al., 2012). The data presented here suggest that climatic variability and favourable physical conditions have contributed to the recent resurgence of

cyanobacterial blooms, although potential changes in nutrient supply cannot be excluded. Our findings indicate that cyanobacterial dynamics therefore appear to reflect the interaction between internal nutrient availability and physical forcing, while the contribution of nutrient transport through diversion wall leakage remains uncertain.

4.3 Comparison with other Lake Rotoiti sites

Okawa Bay consistently exhibits higher chl-a, greater cyanobacterial biovolume, and lower water clarity than deeper Lake Rotoiti basin sites. Persistent differences between Okawa Bay and the main basin despite lake-wide interventions indicate that local morphometry and hydrodynamics exert strong control over ecological responses in the bay. At the same time, bloom occurrence exhibits clear temporal coherence across sites, particularly in years of elevated bloom activity, despite differences in bloom magnitude across sites. This pattern aligns with the concept of spatial synchrony, where ecological dynamics are driven by regional climatic forcing and exhibit similar directions in fluctuations, even with site-specific differences in the overall magnitude (Kent et al., 2007, Vagnon et al. 2024).

Quantifying relative difference between sites over time (e.g., Okawa Bay 90th percentile chl-a relative to main-basin 90th percentile chl-a) would provide a clear metric of sensitivity. If this ratio remains relatively stable across years, it supports the interpretation that Okawa Bay differs primarily in amplitude rather than in fundamental driver identity. A limitation of this approach is the relatively short duration of the recent resurgence in cyanobacterial blooms. The post-2021 record is insufficient to distinguish between short-term climatic variability and a sustained structural increase in amplification. A longer time series will be required before robust inference can be made.

4.4 Relative roles of meteorological forcing, climatic variability, hydrological exchange, and internal processes

The analysis presented in this report indicates that water quality in Okawa Bay reflects interactions of meteorological forcing, climate variability, water exchange with the western basin, and nutrient availability. Meteorological forcing appears particularly important for bloom expression through its influence on mixing and water exchange. This is supported by statistically significant wind effects at upper chl-a quantiles (Figure 10) and changing importance of wind over time (Figure 15). Climatic variability may also contribute to bloom-conducive conditions with SOI significantly related to chl-a at upper quantiles, although the relatively low explanatory power of the models indicates that this is only one component of the observed variability. Hydrological exchange with the western basin has been demonstrated by tracer modelling and historical inflow studies (Gibbs et al. 2003; Hamilton et al., 2009a, b), while our analysis identified relationships between Okawa Bay chl-a and chl-a and nutrient concentrations at Site 3, particularly under non-bloom conditions. However, the absence of strong monthly-scale coupling

between Ōhau Channel nutrient loads and Okawa Bay chl-a suggests that exchange does not dominate bloom expression under contemporary conditions. Given the small, shallow nature of Okawa Bay, coupled with historical nutrient enrichment and likely episodic hypoxic events, sediment nutrient release may provide an additional nutrient source during periods of stratification and low bottom-water oxygen, with subsequent mixing making these nutrients available within the water column. This interplay between nutrient dynamics and external drivers such as as meteorology and hydrology in controlling eutrophication is well documented across the international literature (Duan et al. 2026).

4.5 Evidence for management-related signals

Breakpoint analyses identified shifts in individual variables but did not reveal a single, sustained regime shift coincident with any one intervention. Ōhau Channel diversion in 2008 and sewage reticulation in the mid-2000s likely reduced external nutrient pressure, contributing to the mid-2000s decline in chl-a and bloom frequency, and a slightly later improvement in water clarity in 2007. However, a lack of a breakpoint for DIN, and an increase in DIN:DRP from 2022 onwards indicates inconsistent mechanisms which may be driving variability in nutrient dynamics. In addition, the resurgence of cyanobacterial blooms in recent years indicates that these interventions did not eliminate bloom susceptibility, in the sense that nutrient concentrations and sediment nutrient pools remain sufficient to support rapid phytoplankton growth when physical conditions become favourable.

The analysis of the effects of weed spraying on nutrient dynamics does not provide quantitative evidence that aquatic weed spraying produces sustained, system-wide increases in TP or TN mass at the temporal scale examined. We used whole-bay TN and TP mass for this analysis because the decay model estimates cumulative release of the total N and P contained in the treated macrophyte biomass, allowing us to examine whether potential nutrient release was associated with detectable changes in the water-column nutrient pool. However, this does not constitute a complete nutrient mass balance. Although nutrient release from decaying macrophytes is mechanistically plausible and likely, predicted release based on Scholes (2015a) parameter ranges does not consistently align with observed post-spray trajectories. In many cases, the model predicts cumulative nutrient release that exceeds the observed net change in nutrient mass. This is not unexpected, as observed nutrient mass reflects the combined effects of nutrient release, biological uptake, sedimentation, mixing and exchange with the western basin, processes that are not explicitly represented in the decay model. Importantly, post-event slopes in spray years are not distinguishable from slopes observed in non-spray years during the same seasonal window. This indicates that background seasonal variability and meteorological forcing contribute substantially to nutrient dynamics in Okawa Bay, and represents processes not included in the current modelling approach. Therefore, these results do not imply that nutrient release following

spraying does not occur. Rather, they indicate that any such release is not consistently detectable above natural variability in whole-bay TP and TN mass using monthly monitoring data.

The pre- and post-wall water quality comparisons suggest that the strength of relationships between Okawa Bay chl-a and both external (Ōhau Channel) and internal (Site 3) nutrient indicators varied across the pre-wall, post-wall and post-2020 periods, although not always in the direction hypothesised. The weak pre-wall relationships are not necessarily inconsistent with a mechanistic influence of Ōhau Channel inflows, because Rotorua-derived water can enter Lake Rotoiti as surface flows, interflows, or underflows, and the extent to which these waters influence Okawa Bay depends on depth of insertion, entrainment, and wind-driven redistribution (Vincent et al., 1991; Gibbs 1992). A consistently strong relationship at monthly sampling resolution would therefore not necessarily be expected. Post-2020, the fitted slopes generally appeared steeper, although these differences were not formally tested for statistical significance. The observed patterns cannot be attributed to a particular mechanism based on the available data. Additionally, this analysis treats the post-2020 period as a single interval, although the extent of leakage through the diversion wall has changed over time. This limits our ability to resolve any gradual changes in hydrological connectivity. From a hydrodynamic perspective, increased leakage would alter the delivery of Rotorua-derived water to the western basin, but its subsequent influence on Okawa Bay would depend on the density structure, wind forcing and mixing conditions. However, even with holes in the diversion wall, water entering through them may behave as a density current in Lake Rotoiti. Therefore, changes in meteorological conditions and associated mixing and transport processes could also contribute to the observed differences in statistical relationships. Given the persistent importance of internal nutrient cycling and the bay's distinct hydrodynamic characteristics, these changes should not be interpreted as evidence of a fundamental shift in the processes controlling water quality in Okawa Bay.

Overall, management interventions appear to have modified nutrient inputs and hydrological conditions within the wider Rotorua–Rotoiti system. However, their effects on Okawa Bay are mediated by its connectivity with the western basin, meteorological forcing and internal nutrient cycling. The relative contribution of individual interventions to observed changes in water quality therefore remains difficult to determine.

4.6 Caveats

Monitoring is predominantly monthly and was designed for State of the Environment reporting rather than process-based experimentation. In shallow polymictic embayments, stratification, oxygen depletion, nutrient release, and wind-driven transport can occur over hours to days. Monthly sampling therefore captures system state rather than continuous process dynamics. Non-detection of short-lived weather

and transport events or nutrient pulses should be interpreted as resolution-limited rather than mechanistically definitive.

Connectivity between Okawa Bay and the western basin is inferred from previous tracer modelling, spatial gradients, and wind associations in our statistical analysis. Continuous flow measurements are not available for the study period. Exchange is therefore supported conceptually and based on the model but not continuously quantified under all meteorological conditions.

Water-quality monitoring within Okawa Bay is conducted at a single long-term station, and cyanobacteria monitoring at a nearshore recreational site. While consistent station placement supports interannual comparability, horizontal and vertical heterogeneity within semi-enclosed embayments means that whole-bay averages cannot be resolved fully at all times. Frequency and trend patterns are robust; absolute magnitudes may reflect spatial sampling design.

Evaluation of weed control and diversion effects relies on retrospective observational data rather than controlled experiments. Co-occurring meteorological variability complicates causal attribution. Absence of statistically robust whole-bay responses does not exclude short-lived or localised effects; it indicates that such signals were not resolvable at monthly scale.

Associations between wind, climate indices, and chl-a are statistically significant but have modest predictive power and do not uniquely partition the relative contributions of climate variability, internal nutrient cycling, and hydrological exchange. Conclusions should therefore be interpreted as scale-dependent and conditional rather than deterministic.

4.7 Conclusions

Our results indicate that Okawa Bay is best characterised as a shallow, meteorologically sensitive embayment with ongoing nutrient legacy effects and episodic basin-scale coupling. Cyanobacteria bloom dynamics appear to reflect the interaction between internal nutrient availability and meteorological control of phytoplankton habitat suitability via stratification, mixing, and wind-driven exchange between Okawa Bay and the western basin. Climatic variability may further influence bloom severity, while hydrological exchange with the western basin influences conditions within the bay. Although management interventions have modified nutrient inputs and hydrodynamics in the wider Rotorua–Rotoiti system, their specific contributions to changes in Okawa Bay water quality remain difficult to quantify.

This interpretation is quantitatively consistent with both the analyses presented here and the broader shallow-lake literature. It is also consistent with the early process-based

investigations undertaken in the early 2000s (Ray et al., 2002; Gibbs et al., 2003), which identified dominant sediment nutrient release and substantial exchange flow with the western basin as primary constraints on bloom response to local load reductions. Nutrient–phytoplankton modelling at that time predicted that sewerage reticulation and modest diversion flows would produce only limited short-term improvements in bloom biomass because internal loading and hydrodynamic connectivity would continue to regulate system behaviour. More than two decades later, long-term monitoring indicates modest improvements in some water quality indicators, but continued susceptibility to cyanobacterial blooms. This is consistent with the enduring influence of historical nutrient enrichment and hydrodynamic exchange, which continue to shape the bay's response to management interventions. This provides a proportionate, data-driven basis for management discussion while acknowledging remaining uncertainties and opportunities for targeted further investigation.

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Appendix A

Text A1. Macrophyte decay and nutrient release model

Model description

Nutrient release from sprayed macrophytes was estimated using a first-order decay formulation consistent with Scholes (2015a). The model was applied separately for TP and TN.

For each spray event, the total treated area (A, ha) was obtained from operational records. Biomass density was sampled from literature-supported ranges reported by Scholes (2015a) for North Island lakes (2,940–4,940 kg dry weight ha^{-1}). Biomass density (kg ha^{-1}) was converted to kg m^{-2} and multiplied by treated area (A) to obtain total macrophyte biomass ($M_{biomass}$, kg):

$$M_{biomass} = A \times B$$

where, B is biomass density (kg m^{-2}).

Tissue nutrient content was sampled from Scholes-consistent ranges:

- Phosphorus: 3–10 g P kg^{-1} dry weight
- Nitrogen: 18–36 g N kg^{-1} dry weight

These values were converted to kg P or N kg^{-1} dry weight and used to calculate the initial releasable nutrient pool:

$$M_{0,P} = M_{biomass} \times C_P$$

$$M_{0,N} = M_{biomass} \times C_N$$

Where, C_P and C_N represent tissue nutrient fractions (kg P or N kg^{-1} dry weight). Nutrient release was modelled as a first-order decay process:

$$M_{(t)} = M_0(1 - e^{-kt})$$

Where, $M_{(t)}$ is cumulative nutrient release (kg) at time (t) (days), M_0 is the initial nutrient pool (kg), and k is the decay constant (day^{-1}). The decay constant was defined such that 95% of releasable biomass decomposes within 84 days:

$$k = -\frac{\ln(0.05)}{84}$$

This formulation assumes monotonic release and does not explicitly represent nutrient immobilisation, adsorption, export, or biological uptake.

Monte Carlo uncertainty propagation

To account for uncertainty in biomass density and tissue nutrient content, 10,000 Monte Carlo simulations were performed for each spray event. Biomass density and tissue nutrient content were sampled from uniform distributions bounded by the Scholes (2015a) parameter ranges described above.

For each simulation, cumulative nutrient release was calculated over the 120 day post-spray window. The median and 5th–95th percentile envelopes were retained as the modelled estimate and uncertainty bounds.

Comparson with observed nutrient mass

Because the model estimates cumulative nutrient release rather than absolute whole-bay mass, predicted time-series segments were anchored to the first observed post-spray nutrient mass measurement within each event window.

If t_a represents the first post-spray observation and $M_{obs}(t_a)$ the observed whole-bay nutrient mass, the anchored modelled mass at time (t) was calculated as:

$$M_{model,abs}(t) = M_{obs}(t_a) + [M_{model}(t) - M_{model}(t_a)]$$

This ensures that modelled trajectories are plotted on the same absolute mass scale as observations, while preserving the modelled cumulative release structure.

Event-aligned comparison framework

For each spray year, a 60-day pre-spray window was used to calculate baseline mean nutrient mass. A 120-day post-spray window was used to evaluate change in mass relative to this baseline.

Linear rates of change (kg day^{-1}) were calculated over the post-event window for both observed mass and modelled cumulative release. Spray years were compared to non-spray years (pseudo-aligned to 1 December) to assess whether post-event nutrient trajectories exceeded seasonal background variability.

Model scope and limitations

The macrophyte decay model is intentionally simplified and represents cumulative nutrient release from treated biomass only. It does not explicitly simulate subsequent biogeochemical or hydrodynamic processes, including nutrient adsorption to sediments, particulate settling, microbial immobilisation, denitrification, biological uptake, or export through exchange with the main basin of Lake Rotoiti. Nor does it incorporate explicit oxygen dynamics or redox-dependent sediment phosphorus release. The model therefore estimates potential nutrient input from biomass decay rather than persistent water-column accumulation.

Treated area data were derived from operational management records rather than measurements collected specifically for process modelling. Biomass density and tissue nutrient content were parameterised using literature-based ranges consistent with Scholes (2015a), rather than site-specific post-spray measurements for each year. No contemporaneous measurements of harvested biomass, decay rates, or direct nutrient flux were available to further constrain parameter values. Uncertainty in biomass density and tissue nutrient content was propagated through Monte Carlo simulation; however, the selected ranges are assumed to represent realistic bounds for Okawa Bay conditions.

The model assumes first-order decay kinetics, with approximately 95% of releasable biomass decomposing within 84 days. In practice, decomposition rates may vary with temperature, oxygen availability, plant species composition, and physical disturbance. Faster decomposition would produce short-term peaks that may not be resolved by monthly monitoring, whereas slower decomposition would result in more gradual release. The monitoring dataset therefore limits detection of short-lived pulses.

Area-based scaling further assumes that treated polygons correspond proportionally to biomass affected. In reality, vegetation density may be spatially heterogeneous within treated areas, introducing additional uncertainty in estimated nutrient pools.

Accordingly, modelled release should be interpreted as a plausible upper-bound estimate of potential nutrient input to the water column under the assumed parameter space. Comparison with observed whole-bay nutrient mass trajectories therefore tests whether such release is detectable at the spatial and temporal resolution of the monitoring programme, rather than providing a fully resolved mechanistic simulation of post-spray nutrient processing.

Table A1. Significant dates identified in the breakpoint analysis for Okawa Bay water-quality variables. Variable abbreviations and units are defined in Table 1.

variable	Okawa Bay	Site 3	Site 4
DRP_mgm3_Bottom	2011-02-01	2014-10-01	2006-08-01
DRP_mgm3_Integrated	2011-01-01	2008-05-01	2007-05-01
chla_mgm3_Integrated	2006-07-01	2013-07-01	--
	--	2019-07-01	--
secchi_m_Integrated	2007-03-01	2012-10-01	2012-10-01
	2010-12-01	2017-11-01	--
	2016-04-01	--	--
DN_DP_Integrated	2022-04-01	2006-08-01	2006-12-01
DIN_mgm3_Bottom	--	2014-10-01	2006-11-01
	--	--	2012-05-01
DIN_mgm3_Integrated	--	2006-08-01	2006-12-01
DN_DP_Bottom	--	2007-01-01	2007-09-01
	--	2010-11-01	2012-10-01
	--	--	2015-10-01

Table A2. Summary statistics of key water quality variables across sites in Lake Rotoiti. Variable abbreviations and units are defined in Table 1.

variable	Site	mean	median	max	min	sd
DIN (mg m^{-3})	Okawa Bay	14.6	6.1	456.0	1.0	36.8
	Site 3	21.9	8.0	250.0	-2.0	38.8
	Site 4	21.7	6.0	232.0	1.0	41.7
DIN:DRP	Okawa Bay	7.6	3.0	111.7	0.2	15.0
	Site 3	11.7	6.1	174.7	-4.0	20.7
	Site 4	8.2	3.8	142.6	0.2	15.6
DRP (mg m^{-3})	Okawa Bay	6.3	5.0	29.0	0.0	4.7
	Site 3	6.4	3.1	38.0	0.0	6.7
	Site 4	8.5	5.0	43.0	0.2	8.7
TN:TP	Okawa Bay	23.3	21.3	69.8	9.6	7.9
	Site 3	25.2	23.8	88.1	6.7	11.4
	Site 4	24.1	22.9	97.3	5.0	13.7
Chl-a (µg m^{-3})	Okawa Bay	12.0	7.4	234.0	1.0	19.4
	Site 3	7.1	6.4	29.0	1.3	3.7
	Site 4	5.8	4.8	63.7	1.0	4.8
Secchi (m)	Okawa Bay	3.2	3.1	5.4	0.8	1.0
	Site 3	5.0	5.0	9.2	2.5	1.2
	Site 4	6.2	6.3	10.2	2.0	1.4

Table A3. Summary statistics across seasons for each site in Lake Rotoiti and selected water quality variables. Variable abbreviations and units are defined in Table 1.

Season	Site	Variable	Mean	Median	Max	Min	S.D.
spring	Okawa Bay	DIN (mg m^{-3})	7.4	5.3	52.5	1.0	7.4
spring	Site 3	DIN (mg m^{-3})	17.4	6.1	129.0	0.0	26.1
spring	Site 4	DIN (mg m^{-3})	11.6	5.0	152.0	1.0	27.4
summer	Okawa Bay	DIN (mg m^{-3})	21.2	6.0	456.0	1.0	64.6
summer	Site 3	DIN (mg m^{-3})	13.5	6.1	205.0	-2.0	27.0
summer	Site 4	DIN (mg m^{-3})	14.4	6.3	199.0	1.3	31.0
autumn	Okawa Bay	DIN (mg m^{-3})	10.8	7.5	100.4	1.0	13.7
autumn	Site 3	DIN (mg m^{-3})	21.3	8.8	250.0	1.0	41.7
autumn	Site 4	DIN (mg m^{-3})	17.4	6.1	222.0	1.5	36.1
winter	Okawa Bay	DIN (mg m^{-3})	19.1	10.0	141.0	1.5	27.7
winter	Site 3	DIN (mg m^{-3})	36.7	11.5	215.0	-1.5	52.6
winter	Site 4	DIN (mg m^{-3})	45.3	22.0	232.0	1.0	58.9
spring	Okawa Bay	DN:DP	Inf	2.2	Inf	0.5	NA
spring	Site 3	DN:DP	Inf	3.7	Inf	0.0	NA
spring	Site 4	DN:DP	3.4	1.4	24.9	0.2	4.5
summer	Okawa Bay	DN:DP	6.6	2.6	85.2	0.2	12.6
summer	Site 3	DN:DP	10.9	6.7	56.7	-4.0	12.9
summer	Site 4	DN:DP	11.0	5.9	114.3	1.1	17.2
autumn	Okawa Bay	DN:DP	9.9	3.3	111.7	0.4	20.9
autumn	Site 3	DN:DP	22.3	10.7	174.7	0.7	34.6
autumn	Site 4	DN:DP	13.2	5.4	142.6	1.1	23.4
winter	Okawa Bay	DN:DP	9.3	4.6	77.9	0.4	14.7
winter	Site 3	DN:DP	5.1	2.7	27.9	-0.4	5.8
winter	Site 4	DN:DP	4.5	2.8	16.6	0.2	4.6
spring	Okawa Bay	DRP (mg m^{-3})	6.4	6.4	17.0	0.0	4.0
spring	Site 3	DRP (mg m^{-3})	6.4	6.3	17.0	0.0	4.3
spring	Site 4	DRP (mg m^{-3})	9.7	9.8	43.0	0.8	7.5
summer	Okawa Bay	DRP (mg m^{-3})	8.1	5.9	29.0	0.5	6.5
summer	Site 3	DRP (mg m^{-3})	3.3	3.0	16.1	0.2	3.0
summer	Site 4	DRP (mg m^{-3})	3.4	3.0	12.0	0.2	2.2
autumn	Okawa Bay	DRP (mg m^{-3})	4.9	4.0	21.0	0.5	3.5
autumn	Site 3	DRP (mg m^{-3})	2.5	2.0	9.0	0.5	1.6
autumn	Site 4	DRP (mg m^{-3})	3.3	2.1	10.0	0.5	2.3
winter	Okawa Bay	DRP (mg m^{-3})	5.7	4.9	18.0	1.1	3.7
winter	Site 3	DRP (mg m^{-3})	14.0	14.0	38.0	2.0	8.5

winter	Site 4	DRP (mg m^{-3})	19.1	20.0	41.0	2.0	9.3
spring	Okawa Bay	Chl-a (μg m^{-3})	5.8	4.7	20.2	1.0	3.6
spring	Site 3	Chl-a (μg m^{-3})	5.5	5.0	21.3	1.3	3.1
spring	Site 4	Chl-a (μg m^{-3})	4.9	3.4	63.7	1.0	8.0
summer	Okawa Bay	Chl-a (μg m^{-3})	9.4	6.7	39.6	1.4	8.4
summer	Site 3	Chl-a (μg m^{-3})	6.7	6.2	14.7	1.4	3.0
summer	Site 4	Chl-a (μg m^{-3})	4.2	4.0	11.6	1.0	2.0
autumn	Okawa Bay	Chl-a (μg m^{-3})	21.3	10.8	234.0	1.9	34.2
autumn	Site 3	Chl-a (μg m^{-3})	8.8	7.6	29.0	1.9	4.5
autumn	Site 4	Chl-a (μg m^{-3})	6.7	5.5	16.9	2.7	3.1
winter	Okawa Bay	Chl-a (μg m^{-3})	11.5	9.2	53.0	1.6	9.4
winter	Site 3	Chl-a (μg m^{-3})	7.6	7.5	15.6	1.5	3.0
winter	Site 4	Chl-a (μg m^{-3})	7.3	6.1	18.6	2.3	3.5
spring	Okawa Bay	Secchi depth (m)	3.3	3.2	5.3	1.7	0.9
spring	Site 3	Secchi depth (m)	4.5	4.3	7.7	2.5	1.1
spring	Site 4	Secchi depth (m)	6.1	6.3	9.0	3.3	1.3
summer	Okawa Bay	Secchi depth (m)	3.0	2.9	5.0	1.3	0.9
summer	Site 3	Secchi depth (m)	5.0	4.9	9.2	2.8	1.3
summer	Site 4	Secchi depth (m)	6.8	7.0	9.4	3.3	1.5
autumn	Okawa Bay	Secchi depth (m)	3.2	3.0	5.0	0.8	1.2
autumn	Site 3	Secchi depth (m)	5.3	5.3	8.1	2.7	1.1
autumn	Site 4	Secchi depth (m)	6.3	6.4	10.2	2.0	1.4
winter	Okawa Bay	Secchi depth (m)	3.3	3.1	5.4	1.3	1.1
winter	Site 3	Secchi depth (m)	5.3	5.3	8.1	3.5	1.0
winter	Site 4	Secchi depth (m)	5.6	5.4	8.5	3.8	1.1

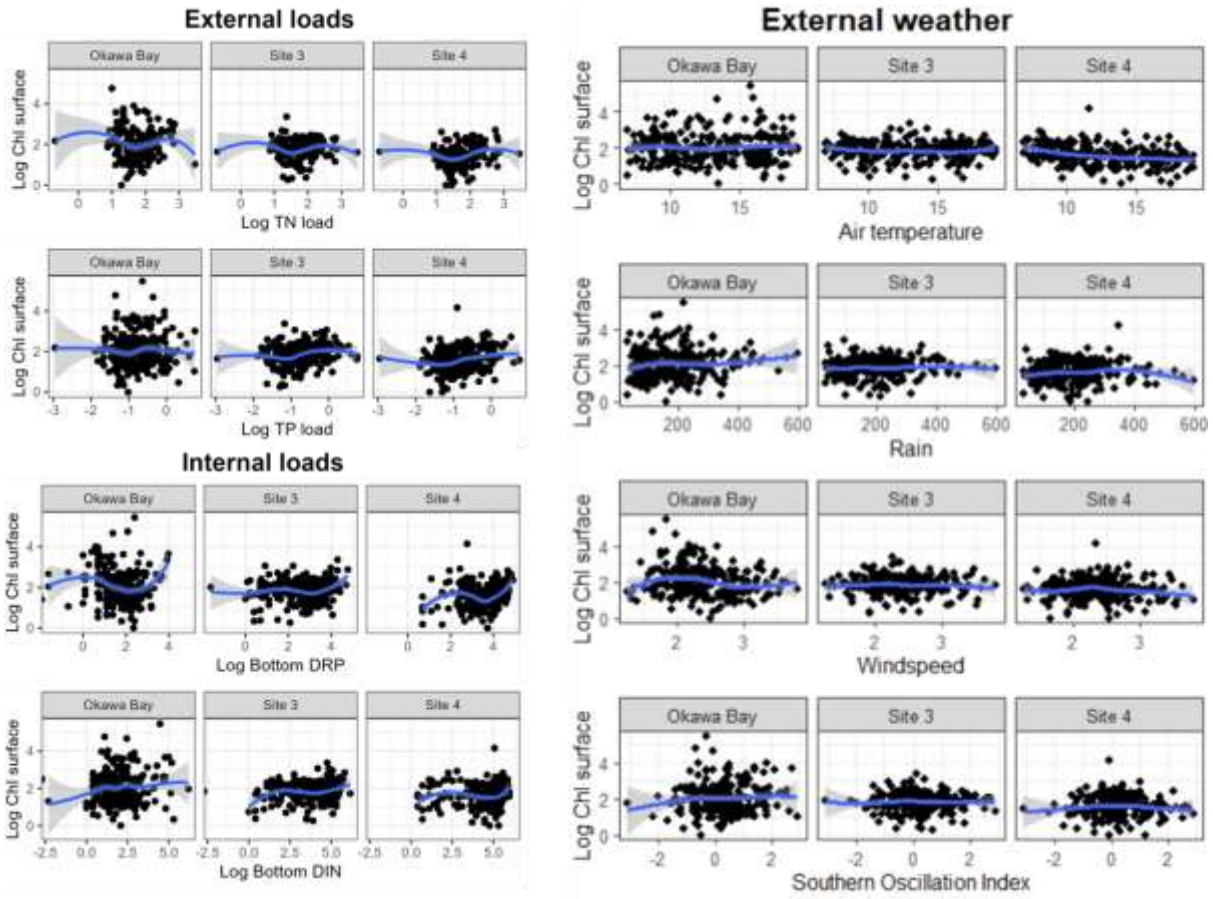


Figure A1. Relationships between common stressors and surface chl-a at three sites in Lake Rotoiti. Note that no total nitrogen data from the Ōhau Channel was available from before the wall (2008-01-01) and only total phosphorus data is compared before and after.

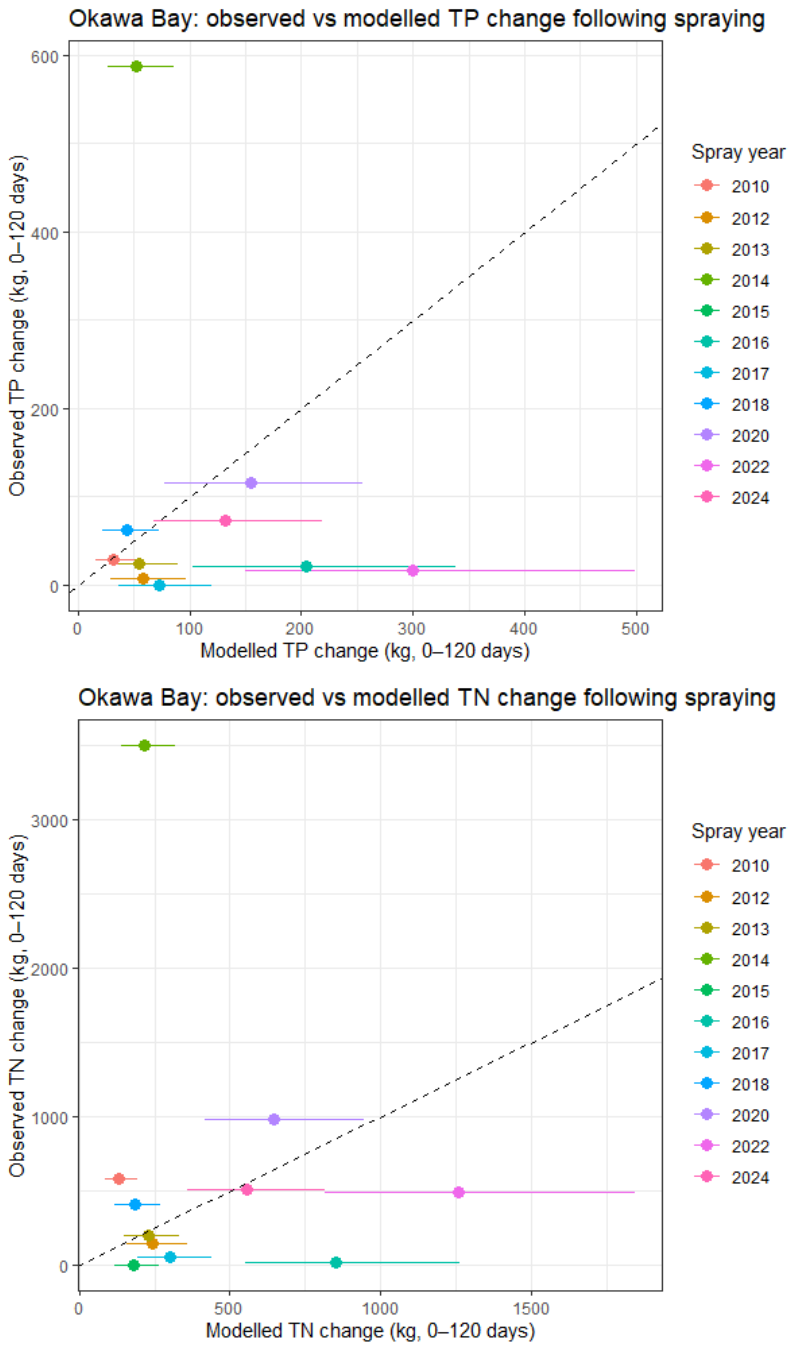


Figure A2: Comparison of modelled and observed change in whole-bay total phosphorus (TP) and total nitrogen (TN) mass over the 120 day post-spray window for spray years in Okawa Bay. Observed change is calculated as the difference between the maximum measured TP mass within the post-event window and the first post-spray observation. Modelled change represents cumulative nutrient release from the macrophyte decay model, with horizontal error bars showing the 5th–95th percentile uncertainty range from Monte Carlo simulations. The dashed line indicates a 1:1 line between modelled and observed change. Points above the line indicate underprediction by the model; points below the line indicate overprediction.

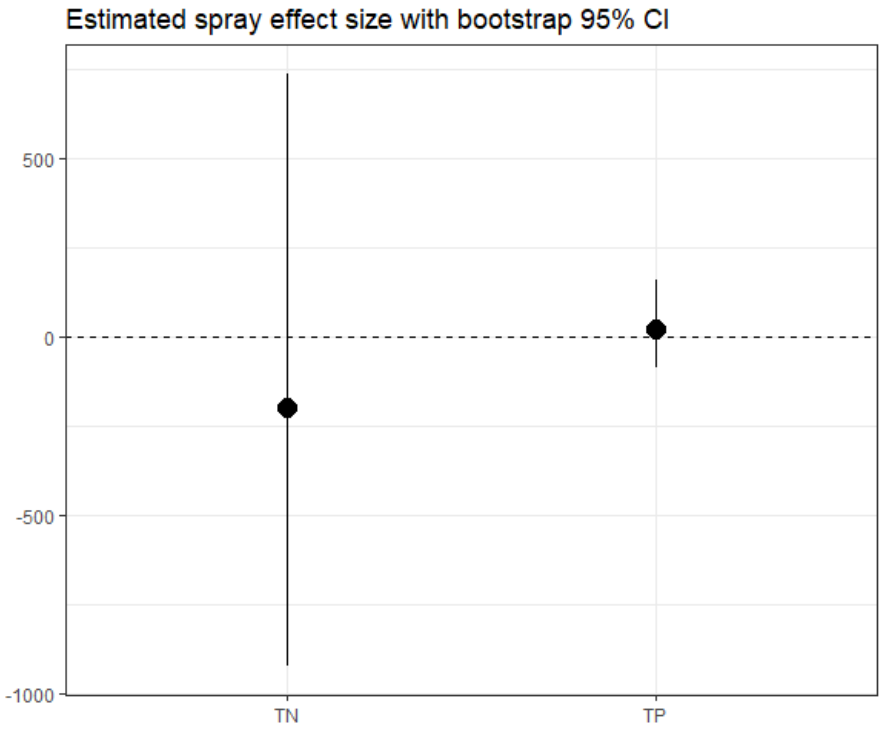


Figure A3. Estimated effect size of aquatic weed spraying on whole-bay nutrient mass change in Okawa Bay over the 120 day post-event window. Points represent the median bootstrap estimate of the mean difference in observed nutrient mass change between spray and non-spray years (Spray − Non-spray), and vertical bars show the 95% confidence interval derived from 10,000 bootstrap resamples. The dashed horizontal line indicates zero difference. Confidence intervals overlapping zero indicate that post-event changes in TP and TN mass during spray years are not statistically distinguishable from seasonal variability observed in non-spray years.