



# Technical Report on Vegetation Status in Waituna Lagoon: 2009–2018



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## Introduction

This technical report accompanies the summary report on vegetation status in Waituna Lagoon in 2018 (de Winton and Mouton 2018). We review the lagoon conditions over the period of vegetation monitoring, including water level, mouth status and duration (Section 1), as well as spatio-temporal environmental patterns from State of the Environment data (Section 2).

As background to the summary report, this technical report provides descriptions of monitoring methods and presents summaries of data and analysis (Sections 3, 4 and 5).

Following the collection of a long-term dataset (2011–2018) it was possible to undertake a spatio-temporal analysis of vegetation patterns (Section 6) to gain a more in-depth understanding of the responses of vegetation. We briefly conclude what the findings mean for managing lagoon openings.

<sup>1</sup> <http://www.legislation.govt.nz/act/public/1998/0097/7.0/DLM431306.html>



# 1. Water Level Regime

## Methods

Water level data supplied by Environment Southland from the gauge at Waghorns Road was examined to identify lagoon openings by the onset of a sudden, substantial reduction in values. Lagoon closure was estimated from timing of subsequent, sustained increases in level. The total time period for openings was calculated, the mouth status was confirmed and the duration of that status before each monitoring event was calculated as months (one month is 30 days).

## Results

At the time of vegetation monitoring in 2018, Waituna Lagoon had been closed to the sea for 13.7 months (Figure 1). Although the lagoon was closed, the water level was lower than previous closed lagoon conditions (2009–10, 2012, 2015–17) (Figure 2). This was because of an extended spring 2017 to summer 2018 drought in Southland that preceded the 2018 monitoring. Nevertheless, in 2018 the water level was higher than during the previous monitoring occasions when the Lagoon was open to the sea (2011, 2013–14).

Prior to 2018, periods of closure in Waituna Lagoon have ranged from 1 to 6.2 months before monitoring. The lagoon has also been open for 3.9 to 6.2 months prior to three monitoring occasions (Figure 1). The years that exceeded three months of lagoon closure before a monitoring event were 2009, 2010, 2012, 2015, 2016 and 2018.

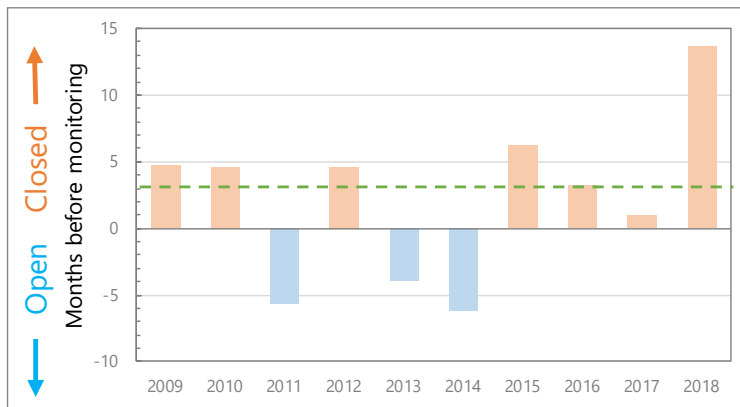


Figure 1: Diverging barplot showing the number of months for which Waituna Lagoon was open or closed prior to monitoring (as indicated by the y axis). The dotted line indicates the ecological target of three months of lagoon closure before monitoring.

## Discussion

The 2018 monitoring event follows an extended closed period for the lagoon. In the following numbered headings, the influence of lagoon openings on physico-chemical conditions and vegetation development are described.

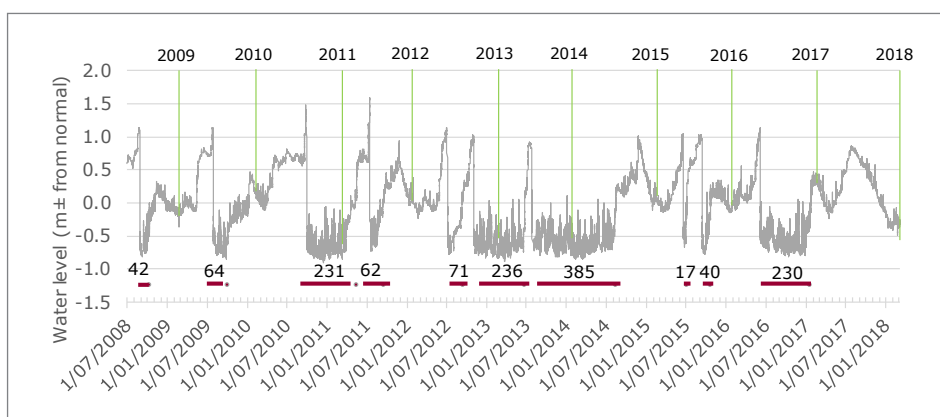


Figure 2: Plot showing the continuous water level time series for Waituna Lagoon, measured at Waghorns Road. Periods of lagoon opening are indicated by horizontal red lines. The number of days during which the lagoon was open correspond to the numbers on top of the horizontal red lines. Finally, the date of the annual summer vegetation monitoring events are indicated by green vertical lines.



## 2. SoE Monitoring Data

### Methods

Water quality monitoring data was obtained from Environment Southland's State of Environment monitoring network for Waituna Lagoon from 2011 to 2018. Data from four sampling sites (strategically distributed in the lagoon from East to West and North to South) were obtained and the following eight parameters used, since these were the only parameters continuously monitored between 2011 and 2018:

- Chlorophyll a ( $\text{mg l}^{-1}$ )
- Conductivity ( $\mu\text{S cm}^{-1}$ )
- Salinity (PSU)
- Total Nitrogen (TN,  $\text{g m}^{-3}$ )
- Total phosphorus (TP,  $\text{g m}^{-3}$ )
- Total Suspended Solids (TSS)
- Turbidity (NTU)
- Temperature ( $^{\circ}\text{C}$ )

To visualise patterns in water-quality between seasons and mouth status (open or closed), we used a between-class (Chessel et al. 2004) principal component analysis (PCA, Pearson 1901). Prior to analyses, the examination of a within-year PCA, suggested poor effects of years on water-quality patterns, we therefore analysed all data together. Between-class PCA was computed using the *dudi.pca()* and *bca()* function from the *ade4* (Dray and Dufour 2007) package in R (R Core Team, 2018).



### Results

In the PCA, the difference between seasons and lagoon mouth status explained 24.2% of the variation in water quality data. In addition, most of the variation in water quality between-classes was explained by the first two axes (62 and 28% for axis 1 and 2 respectively (Figure 3A, page 8).

There were clear differences in water quality between seasons and with lagoon mouth status. As indicated by the ordination (Figure 3B), Winter samples (Figure 3B) were opposed to Summer samples (Figure 3B), and intermediate seasons (Spring and Autumn) located towards the center of the plot. The water quality samples also showed differences between mouth status, with samples collected during opening events (purple ellipses) opposed to those collected during closed events (green ellipses).

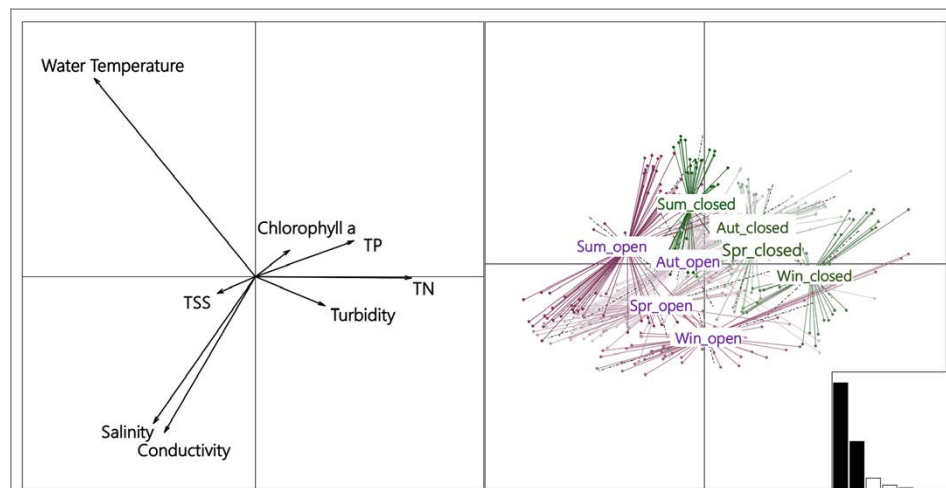


Figure 3: Principal Component Analysis of monthly water quality samples, collected between 2011 to 2018 at four sites in Waituna Lagoon and for eight parameters. The first and the second axis are shown and explain 62% and 28% of the variation in water quality respectively. Figure 3A (left) shows the PCA scores for the seven water quality parameters and Figure 3B (right) shows samples scores for each monitoring event. Site scores are grouped by season and lagoon mouth status, with opening events in purple and closed events in green. Text inside the boxes indicate season and lagoon status (i.e., Sum\_open = Summer and open mouth; Win\_closed = Winter and closed mouth etc.). The barplot at the bottom right of the figure illustrates the variation explained by each axis.

Eutrophication (TN, TP and Chlorophyll *a* concentrations) was much greater during closed Winter periods (Figure 3A), whereas salinity and conductivity were much higher when the lagoon was open (Figure 3A). Finally, the PCA analysis also indicated greater water-column turbidity during winter and much warmer conditions during Summer periods (Figure 3A). The patterns identified by this analysis can serve to predict broad-scale water quality patterns in Waituna Lagoon between seasons and mouth status (open or closed).

## Discussion

Based on our analysis we expect that lagoon closure preceding the 2018 monitoring would have lowered salinity and total suspended solids, but increased temperature and nutrients compared to a lagoon opening. This analysis confirms previous findings of the influence of mouth status on physico-chemical conditions in Waituna Lagoon. A closed Lagoon is associated with higher eutrophication risk (Hodson 2017, Schallenberg and Tyrell 2005, Schallenberg et al. 2010) and higher temperature (Schallenberg and Tyrell 2005) than when it is open.

Our analysis also highlighted a significant seasonal influence on physico-chemical conditions in the lagoon, that interacted with mouth status and would determine the environment leading up to the annual summer vegetation monitoring. The finding of a higher level of nutrients in winter is in keeping with an earlier positive correlation found between water level and TN (Schallenberg and Tyrell, 2005). Higher winter nutrient levels may result from greater run-off due to rainfall events and consequent flooding. Seasonally higher nutrient concentrations during winter may also contrast with summer due to a greater biological uptake in the warmer months.





### 3. Annual Physico-chemical Monitoring

#### Methods

The location of 47-48 monitoring sites is given in Figure 4. (One site could not be sampled due to the migration of the coastal spit in 2014).

At each monitoring site, measurements were made over 2009 to 2018 of:

- Water depth (m)
- Visual clarity as black disk distance (m)

A calibrated multi-sensor meter (Horiba or YSI Exo 1) measured parameters at the water surface and bottom (where depth allowed) that included:

- Temperature (°C)
- Dissolved oxygen (DO, mg l<sup>-1</sup>)
- Salinity (PSU)
- Turbidity (NTU)

Black disk, DO and turbidity were measured from 2011 to 2018 only.



Figure 4: Monitoring sites in Waituna Lagoon. Transects are numbered from 1 to 10 from East to West. The numbers of each transect was allocated on ascending order from North to South.

## Results

Waituna Lagoon was closed for over a year prior to annual summer monitoring in 2018. Consequently, salinity levels in 2018 were low (<2.5 PSU), especially compared to an open lagoon conditions in 2011 (average 17.7 PSU) and 2014 (average 29.1 PSU) that had lasted approximately six months (Figure 5). Salinity was also high in 2017, probably as the lagoon had only been closed for a month before monitoring.

Water depth in 2018 was low due to drought conditions in Southland (Figure 5), with water level below normal (-0.31 m) at the time of monitoring (Figure 2). Nevertheless, lower depths were recorded when the lagoon was open in 2011, 2013 and 2014. Water temperature (average 17.3°C) was comparable to the previous five years but was higher than values recorded over 2009 to 2012 (Figure 5).

In 2018, DO averaged 10.1 mg l<sup>-1</sup> (Figure 6) and all readings exceeded 8 mg l<sup>-1</sup>, indicating the lagoon to be well oxygenated.

Turbidity in 2018 averaged 6.9 NTU and was similar to, or slightly lower than, previous years that all averaged <30 NTU (Figure 6). A common feature across years are the upper outliers that may represent wind-wave disturbance of sediment at some shorelines, but with a maximum of 39 NTU recorded in 2018.

Black disk measurements indicated water clarity in 2018 was at the lower end of previous measurements, similar to 2011 and 2012 but lower than the previous three years from 2015 to 2017 (Figure 6). Given the turbidity results, lower clarity is likely to indicate greater humic ('tea') staining of the lagoon in summer 2018.



## Discussion

Physico-chemical conditions at the time of the annual vegetation monitoring reflect whether the lagoon was open or closed at the time of monitoring and recent mouth status. In 2018, the lagoon recorded low values for salinity and turbidity, as would be expected based on the analysis of water quality patterns in relation to lagoon mouth status and season (Figure 3). However, temperature in 2018 was not elevated compared to the previous open lagoon conditions (2013 and 2014).

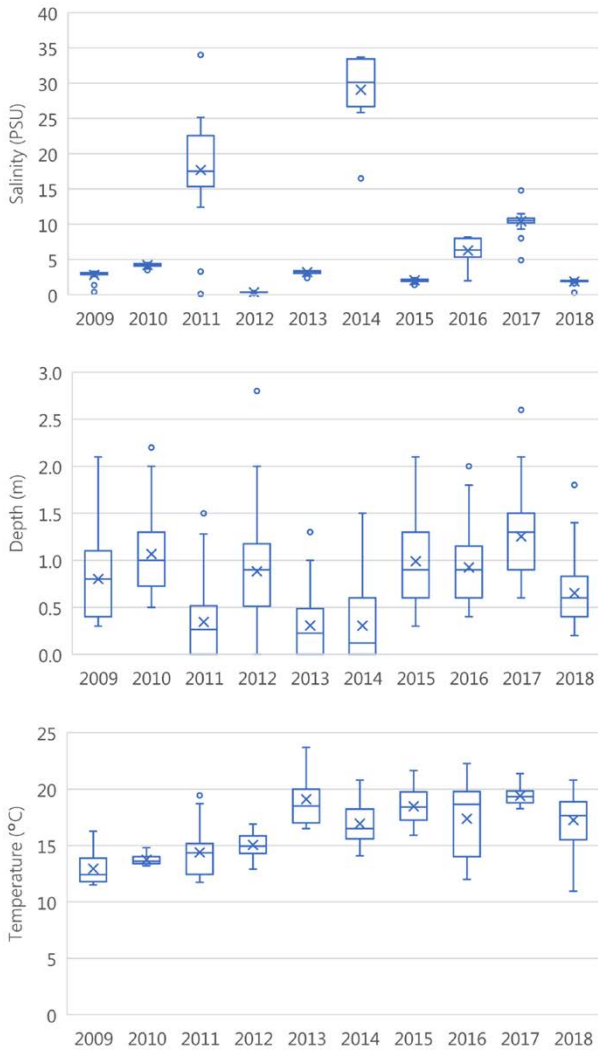


Figure 5: Box and whisker plots of salinity (top), depth (middle) and temperature (bottom) over all monitoring years. (n= 48 or 47).

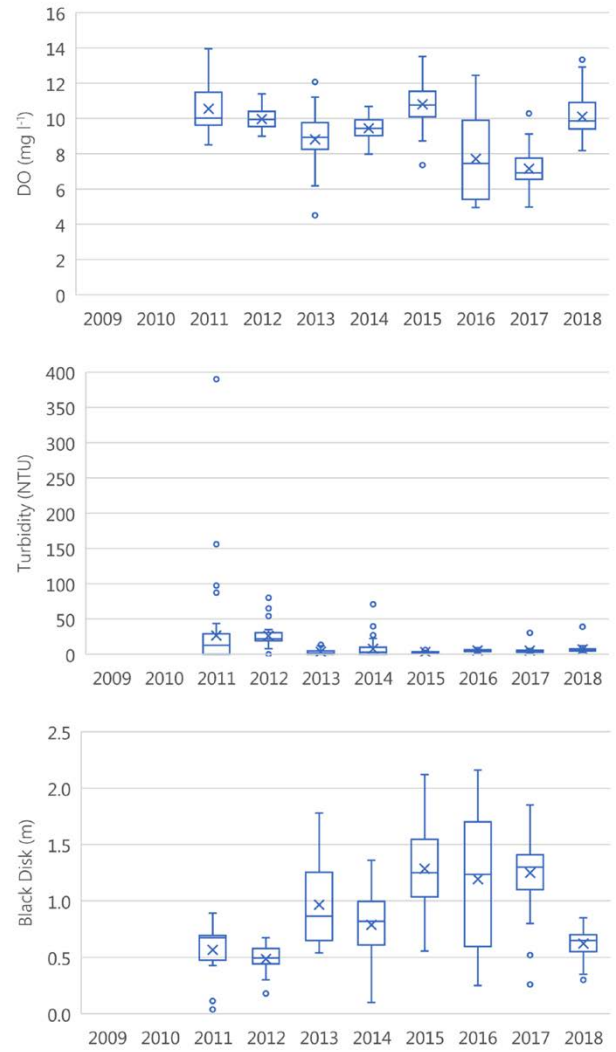
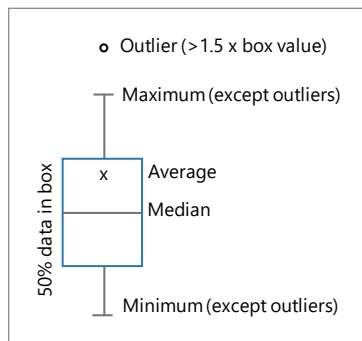


Figure 6: Box and whisker plots of DO (top), turbidity (middle) and black disk (bottom) at the monitoring sites (n= 48 or 47), from 2011 to 2018.



The legend shows features that are plotted on the graphs above.



## 4. Sediment Characteristics

### Methods

At each monitoring site (Figure 4), four replicate samples 15 x 15 cm and 6 cm deep were cut from the lake bed, using a flat based garden hoe, and carefully lifted to the surface. Each sample was assessed for:

- substrate type, (described as combinations of soft or firm mud, sand and gravel) was assigned a score from 1 to 10 describing increasing hardness
- depth (cm) to a blackened layer in the substrate, which indicates sulphide accumulation (elsewhere referred to as the redox potential discontinuity layer, Stevens and Robertson 2007). Depth was categorised into five classes: surface, >0–2, 2–4, >4 cm and layer not recorded.

### Results

Assessed sediment composition in 2018 was intermediate to previous years, with a larger proportion of soft substrates than 2012 to 2017, but an increased proportion of harder substrates than it was assessed in 2009–2011 (Figure 7).

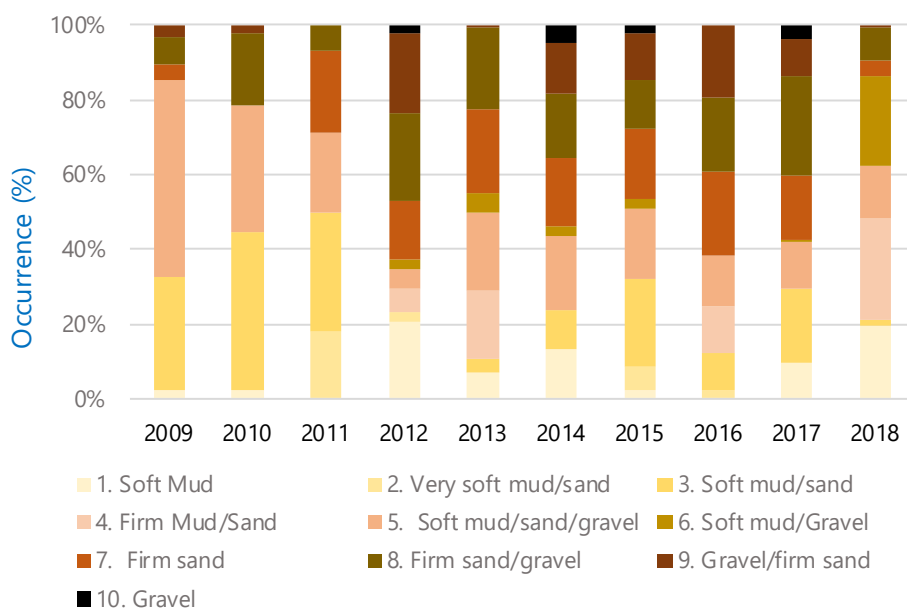


Figure 7: Barplot illustrating the composition in substrate type (% occurrence), recorded during each of the annual monitoring surveys. Substrate types are numbered from softer to harder.





In 2018, the sediment depth to a blackened layer was mainly positioned below the surface and down to almost 2 cm depth, although no blackened layer was present at the sediment surface (Figure 8). Previously, 50% of records have been present at, or near the sediment surface in 2009–2011, 2013 and 2017.

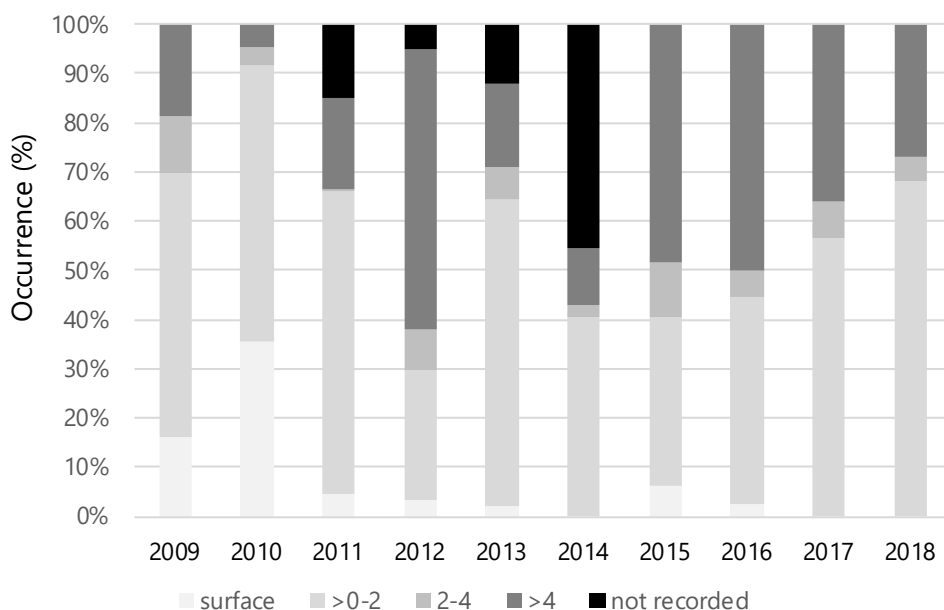


Figure 8: Substrate depth to a blackened layer shown as occurrence (% records) for five incremental depth categories.

## Discussion

Changes in substrate type within Waituna Lagoon towards more widespread soft mud habitat are seen as a deterioration in ecological condition (Stevens and Robertson 2007). Softer substrates were more prevalent over 2009 to 2011, and again in 2018.

The sediment depth to blackened layer may indicate where substrates have lower oxygen status. A low oxygen status can be associated with nutrient release from sediments or the build-up of substances that are toxic for plant growth. More recently, over 2017 and 2018, there appears to have been shallowing of the blackened layer, but not to the extent recorded in 2009–2010.



## 5. Vegetation Development

### Methods

At each site, four replicate samples 15 x 15 cm and 6 cm deep were cut from the sediment, using a flat based garden hoe, and carefully lifted to the surface. Each sample was assessed for:

- Presence of submerged plant species and/or macroalgae types and their cover as %. Where covers were previously recorded as a cover score range<sup>1</sup> in 2009 and 2010, these were translated to a mid-point value.
- Height of each macrophyte species present (cm). Where heights were previously recorded as a range<sup>2</sup> in 2009 and 2010, these were translated to maximum value of the range.
- Life stage of *Ruppia* spp. (vegetative, flowering or post flowering).

Cover and height of *Ruppia* was averaged across the four replicates at each site. Biomass index for *Ruppia* was calculated as the product of average cover and height at each site.

From 2013 onward, macrophyte observations were made at each site by snorkel/SCUBA diver within a circular area of 10 m diameter. The maximum and average cover scores and height were recorded for each macrophyte species and macroalgae present.

### Results

#### Vegetation composition

In 2018, all sites recorded vegetation (Figure 9), with the most frequently encountered plants in the hoe samples being *Ruppia polycarpa* (45 sites), *R. megacarpa* (14 sites), *Myriophyllum triphyllum* (20 sites) and the charophyte *Lamprothamnium macropogon* (10 sites). Macroalgae were a small component of the vegetation with *Ulva intestinalis* being the most common (Figure 9).

Previously, numerous sites without plants were recorded when the lagoon had been open for c. 4-6 months, such as in 2011 (29 sites), 2013 (26 sites) and 2014 (32 sites).

*Ruppia* species have been the most frequently encountered submerged plants in hoe samples over all sampling years, with *R. polycarpa* more common than *R. megacarpa* except in 2011 (Figure 9). Occasionally the two *Ruppia* species occurred in combination and could not be accurately distinguished (*Ruppia* sp. in Figure 9). The freshwater species *Myriophyllum triphyllum* contributed more to vegetation composition in 2009–10 and 2018. The charophyte *Lamprothamnium macropogon* has been conspicuous in contributing to vegetation over the last four years of monitoring as well as 2009 and 2012 (Figure 9).

Macroalgae were much more frequently encountered in hoe samples over the previous three years (2015–2017) than in 2018, with *Ulva intestinalis* and/or filamentous green algae (dominated by *Cladophora* spp.) common in most years (Figure 9). However, the diatom *Bachelotia antillarum* was the most commonly encountered macroalgal type in 2014 (Figure 9). *Bachelotia* was also considered widespread in 2009 and 2010 but was not successfully captured in the samples (Robertson and Stevens 2009, Stevens and Robertson 2010), therefore, cannot be plotted.

<sup>1</sup> 1 = 1–5%, 2 = 5–10%, 3 = 10–20%, 4 = 20–50%, 5 = 50–80%, 6 = 80–100%.

<sup>2</sup> <5 cm, 5–15 cm, 15–30 cm, 30–50 cm, 50–80 cm, 80–100 cm.

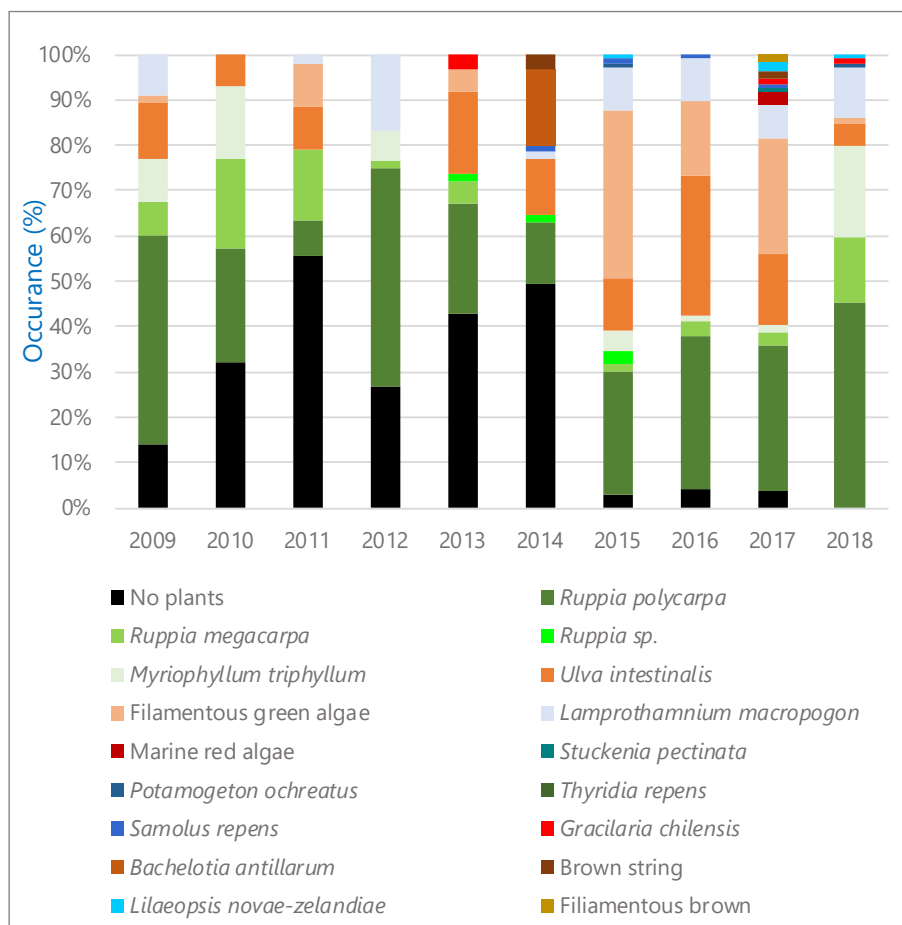


Figure 9: Vegetation composition shown as relative frequency of occurrence (sites recorded) for species or vegetation groups.

### Ruppia abundance

The average cover of *Ruppia* sampled by the hoe method in 2018 was 26% (Figure 10). This lagoon-wide value was lower than that recorded in 2016 (40%), similar to 2009 and 2015, but higher than years 2011–14 and 2017. The less variable *Ruppia* cover values in 2018 reflects their presence at all sites and tendency for more uniform cover (Figure 10, page 16). Twelve sites exceeded 30% cover and two sites, which were both occupied by *R. megacarpa*, had  $\geq 80\%$  cover in 2018. The tendency for *R. megacarpa* to be associated with high covers is seen by its presence at 58% of the sites (23 sites) averaging *Ruppia* covers  $\geq 80\%$  over all monitoring years.

Corresponding *Ruppia* covers at sites using diver observations correlated with the results from the hoe method (Figure 11, page 17). However, observations tended to give higher average covers than the hoe method, particularly where hoe covers were low (Figure 11). Average covers for the lagoon exceeded 30% for the years 2015, 2016 and 2018 using the diver observation method.

Average height of *Ruppia* sampled by the hoe method in 2018 was similar to 2015 and higher than other monitoring years (Figure 10), due partially to the lagoon-wide presence of *Ruppia*. Again, the 13 sites over all monitoring years that had higher average height of  $\geq 75$  cm were associated with *R. megacarpa* (85%), either alone or in combination with *R. polycarpa*. *Ruppia* heights estimated by diver tended to be greater than those based on hoe samples (Figure 11).



'Biomass index' is calculated as the product of average cover times height at sites using the hoe method and is a proxy for biomass in *Ruppia*. In 2018, average biomass index lagoon-wide was similar to 2015 and 2016 and higher than the other monitoring years (Figure 10). High outliers of  $\geq 3000$  biomass index comprised 21 sites across all years that were disproportionately represented by *R. megacarpa* (67%).

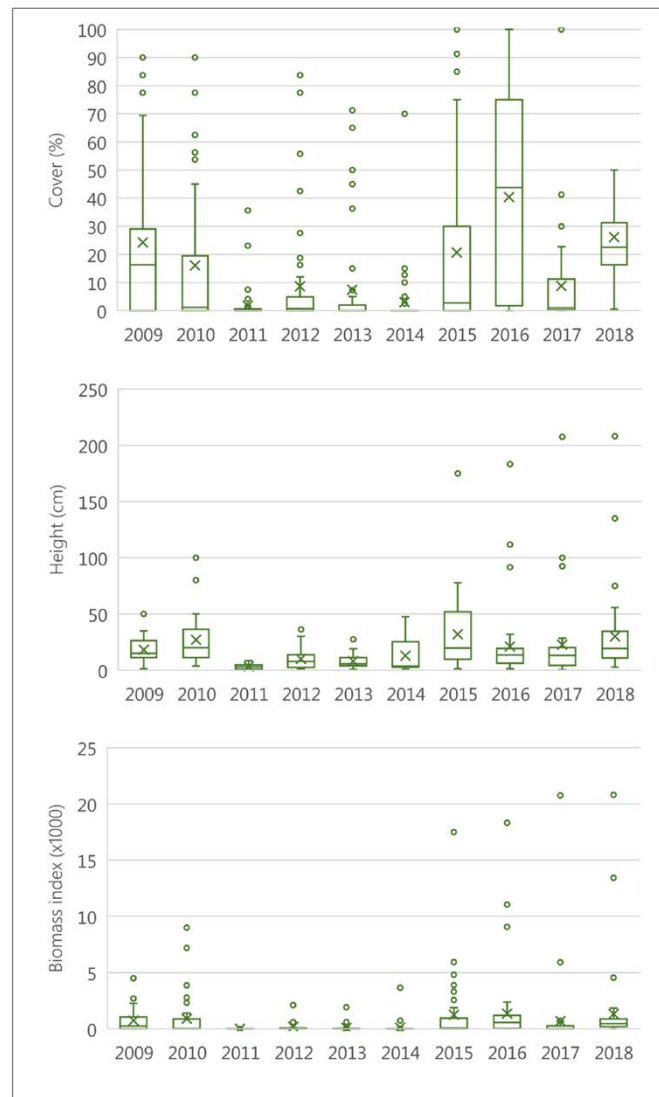


Figure 10: Box and whisker plots of *Ruppia* cover (top), height (middle) and derived biomass index (bottom) over monitoring years. Average of measurements at monitoring sites ( $n= 48$  or  $47$ ).



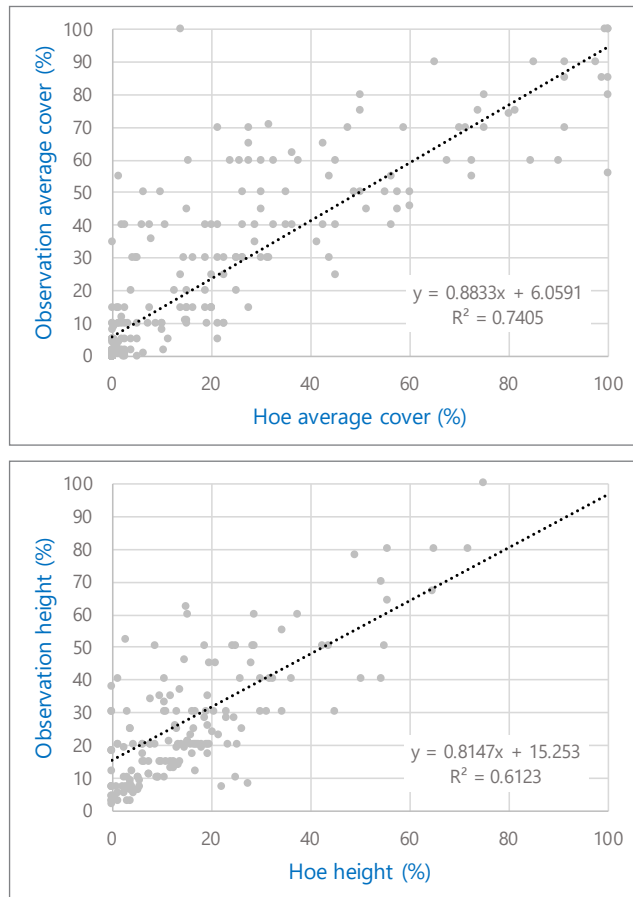


Figure 11: Relationship between *Ruppia* cover (top) and height (bottom) estimated from hoe samples and diver observations within a 10 m diameter area at each site.

### *Ruppia* life-stage

During the 2018 monitoring, 56% of *Ruppia* samples were in a vegetative state and 45% were in a flowering and post flowering state (Figure 12). As in 2018, greater reproduction (flowering or post flowering) was evident for those years where the lagoon was closed for three months or more before monitoring (2009–10, 2012, 2015–16).

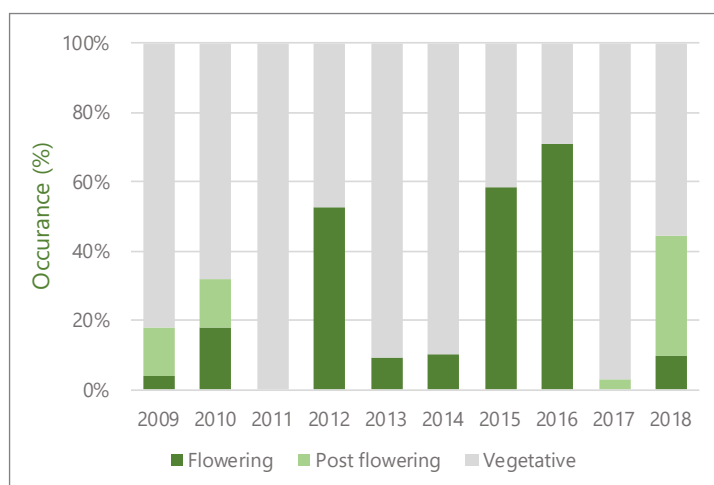
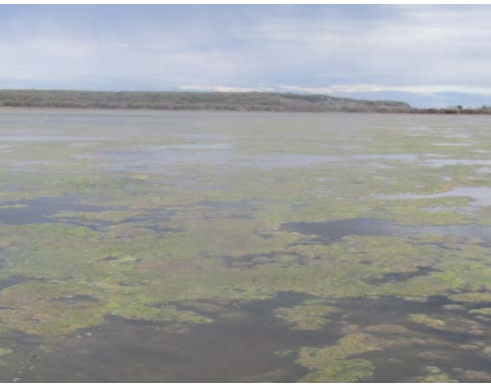


Figure 12: Life-stage category of *Ruppia* species across monitoring years as a proportion of records.



Macroalgae beds can 'lift-off' and grow as a surface mat in still, warm weather.

## Macroalgal cover

Overall cover by macroalgae (excluding minor seaweeds) in 2018 using the hoe method was low, with conspicuous growths only recorded at five sites (Figure 13). This macroalgal abundance was much lower than almost all monitoring years except 2011. In contrast, the years 2015–17 had conspicuous development of macroalgae that exceeded 100% cover at some sites due to combinations of both benthic, epiphytic, and floating mat growths at some sites (Figure 13).

There was a weak correlation between macroalgal cover estimate by hoe samples and overall covers observed by divers (Figure 14). It appears that macroalgal covers are likely to be underestimated by the hoe method, as was reported in 2009 and 2010 (Robertson and Stevens 2009, Stevens and Robertson 2010). Also, macroalgae can occur as benthic, epiphytic and floating growths, which poses problems for the benthic, hoe sampling method. However, observations confirmed macroalgae were not under-estimated in 2018.

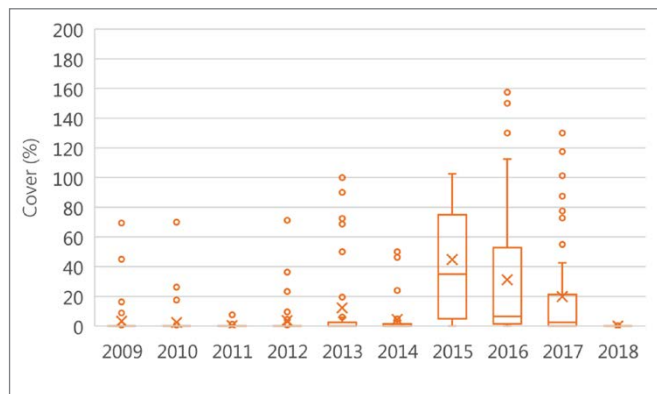
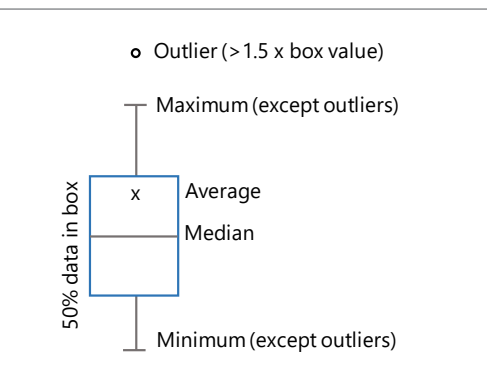


Figure 13: Box and whisker plots of macroalgal cover over monitoring years as an average of measurements at monitoring sites (n= 48 or 47).

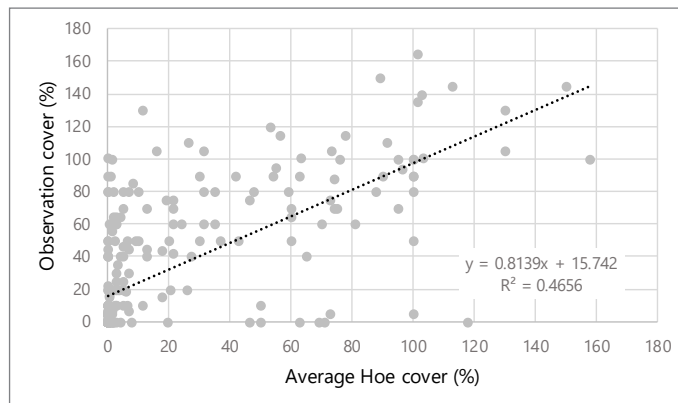


Figure 14: Relationship between macro-algal percentage cover estimated from hoe samples and diver observations within a 10 m diameter area at each site.

## Discussion

Annual monitoring in 2018 confirmed a widespread *Ruppia* vegetation after a long lagoon closure, although cover was not as high as previously recorded apart from some sites with *R. megacarpa*. Unusually, 2018 represented a closed lagoon with very low macroalgal development. Possibly the spring-summer drought in Southland resulted in reduced loads of nutrients entering the lagoon to fuel macroalgal growth or alternatively, blooms of cyanobacteria that occurred in Spring 2017 outcompeted the macroalgae.

A widespread *Ruppia* flowering event was confirmed in 2018, but earlier monitoring results suggest that lagoon openings reduced the extent of flowering. The effect of spring-summer openings on flowering may be via a lower *Ruppia* development as above-ground biomass has been positively correlated to the flower abundance in a *Ruppia* species previously (Santamaría et al. 1995), although not the timing of flowering. It is also likely that plant height plays a role as *Ruppia* flowers must attain the surface for successful pollination. Lagoon openings might also mediate flowering via salinity. Salinity increasing over the range of 6 to 100 PSU (as ppt) led to a decrease in the number of inflorescences and fertile plants produced by *R. polycarpa* in brackish Australian wetlands, although, flowering still occurred at salinities of 40 to 50 g/L (Sim et al. 2006). Also, flowering of *R. megacarpa* corresponded with a fall in salinity in the Australian Coorong system (Geddes 1987). Temperature had a major role influencing timing of flowering in *Ruppia* (Santamaría et al. (1995).



# 6. Spatio-temporal Vegetation Patterns

## Methods

### Biological and environmental data

Ordination statistics and regression models were used to explore the relationship between % cover of each vegetation component measured using the hoe method, (Section 5) and environmental parameters sampled during summer monitoring events between 2011 and 2018 (Section 3 and 4). The data was screened, and identified taxa groups kept (hereafter referred to as species; Table 1).

Environmental parameters measured included substrate hardness, sulphide depth, water depth and five physico-chemical parameters (turbidity, salinity, DO, and water temperature). Black disk was strongly highly correlated with turbidity (Spearman  $r > 0.9$ ) and so was omitted. As surface and bottom water measurements were intercorrelated (Appendix 3), we used the more available surface water measurements for further analyses following Zuur et al. (2010). In addition, 11 samples had no records since they corresponded to dry sites, due to varying lagoon water levels over the years. These were all discarded for further analyses because first, samples without biological records cannot be included in multivariate ordination models and second, predicting the presence or absence of plant species was not the objective of the study, but rather the abundance.

Table 1: Latin or common name and code for each the species used in this study.

Name (latin or common)	Code
<i>Bachelotia</i> sp.	Bac.sp
Brown filament	Bro.fil
Macroalgae dominated by <i>Cladophora</i> spp.	Cld.sp
<i>Erythranthe (Mimulus) cf. repens</i>	Ery.rep
<i>Gracilaria chilensis</i>	Gra.chi
<i>Lamprothamnium macropogon</i>	Lam.mac
<i>Lilaeopsis novae-zelandiae</i>	Lil.nz
<i>Myriophyllum triphyllum</i>	Myr.tri
<i>Potamogeton ochreatus</i>	Pot.och
<i>Ruppia megacarpa</i>	Rup.meg
<i>Ruppia polycarpa</i>	Rup.pol
<i>Samolus repens</i>	Sam.rep
<i>Stuckenia pectinata</i>	Stu.pec
<i>Ulva intestinalis</i>	Ulv.int



## Statistical analyses

### Vegetation patterns and ecological preferences

We employed a co-inertia analysis (COIA; Chessel and Hanafi 1996) to examine the relationship between vegetation community structure and the environmental parameters. COIA is a two-step multivariate ordination procedure, which assesses the linear joint structure between two multidimensional spaces. First a PCA analysis was conducted on the environmental data, and second, a correspondence analysis (CA, Benzécri 1973) was computed on the biological data. The rows of the CA were constrained by those of the PCA. The overall significance of this relationship was tested using a global Monte-Carlo test of the table rows of the PCA and those of the CA.

Because ordination cannot deal with missing values, the missing values for the environmental data (2.4% of the entire dataset) were predicted and imputed to the dataset using a regularised algorithm (the *estimpca()* function in the *missMDA* package; (Josse and Husson 2016). Prior to this analysis, environmental variables were all standardised to their mean and one unit variance, following Legendre and Legendre (2012). COIA was computed using the *dudi.coia()* function in the *ade4* package (Dray and Dufour 2007) in R software (R Core Team, 2018).

### Ecological threshold response curves

To further characterise and understand the distribution and ecological responses of plant species to environmental factors in Waituna Lagoon, we used boosted regression tree models (BRT; Ridgeway 2006, Elith & Leathwick 2017) to characterise the ecological response curves of individual species to the seven environmental parameters. Here, only data for four vegetation components (*R. polycarpa*, *R. megacarpa*, and *U. intestinalis* and macroalgae dominated by *Cladophora* spp.) had sufficient samples for models to be computed. BRT are machine-learning models, that are based on classification and regression tree analysis (CART; Breiman 1984) but use boosting to optimise predictive performances (Ridgeway 2006).

BRT models characterize breakpoints (also referred to as “splits”) in individual relationships between a set of predictors (environmental parameters) and a response variable. These relationships, can be visualised using partial-dependence plots. A partial dependence plot is the relationship between a predictor and a response, when each of the other predictors are held constant (Elith et al. 2008). Here, all predictors were held at their mean values following (Elith et al. 2008). According to Clapcott et al. (2010), partial dependence plots can be used to characterise ecological threshold response curves.

Because BRT models require the response variables to meet assumptions of normality, the abundances of the four vegetation components were Box-Cox transformed prior to analyses, and models fitted assuming Gaussian error-type distributions. The rest of the tuning parameters for every model were set following Elith et al. (2008). We computed BRT models using the *gbm.step()* function in the *dismo* package (Hijmans et al. 2017) in R (R Core Team, 2018).

## Results

### Vegetation patterns and ecological preferences

The global Monte-Carlo test indicated a significant association between vegetation community structure and environmental parameters ( $p < 0.05$ ), and COAI indicated a fair association fit ( $R_v = 0.16$ ). The first two axes (Figure 15) respectively explained 32% and 15% of the variation in community structure.

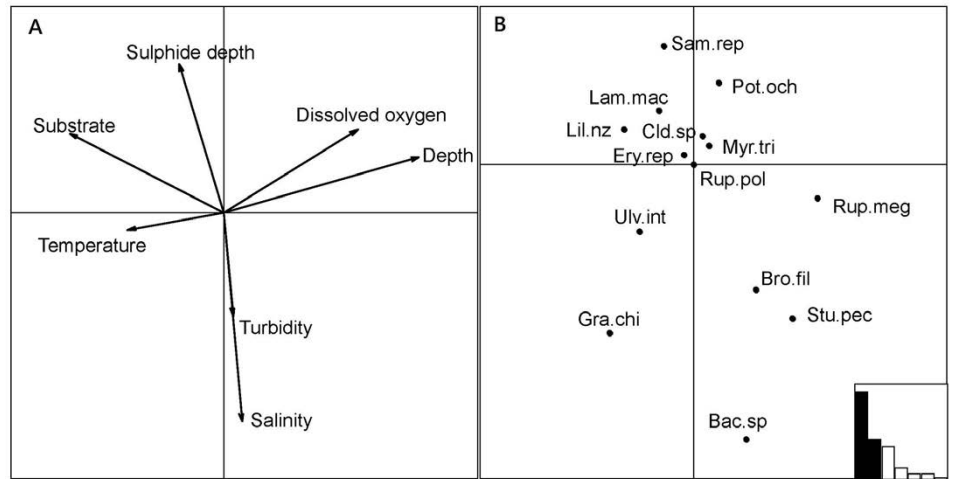


Figure 15A and 15B: Co-inertia (COIA) analysis of vegetation community structure in relation to seven environmental parameters sampled at 47-48 sites in Waituna Lagoon, between 2011 to 2018. The first and the second axis are shown and explain 32% and 15% of the variation in water quality respectively. The Figure 15A shows the COIA scores for the seven environmental parameters and the Figure 15B shows the COIA scores for each species. For species codes see Table 1, page 20. Bar plot at the bottom right of the figure illustrates the variation explained by each axis.

On the ordination of the environmental variables (Figure 15A), the first axis was positively correlated (Figure 15B) to depth and dissolved oxygen, and negatively correlated to temperature and substrate hardness. Salinity and turbidity were negatively correlated to the second axis (bottom of Figure 15A) but sulphide depth was positively correlated. The first axis, therefore, describes a depth gradient in the lagoon, with greater oxygen concentrations and cooler waters. Whereas the second axis is mainly related to sea water entrance.

The position of the species on the ordination (Figure 15B), can inform on their major ecological preferences. For example, *R. megacarpa* is associated with greater depth and dissolved oxygen concentrations, cooler temperatures and softer substrate, whereas *R. polycarpa*, is associated with more universal conditions. Furthermore, the y-axis separated estuarine plant species (*Gracilaria chilensis*, *Ulva intestinalis* and *Bachelotia antillarum*), which preferred higher salinity than freshwater plants (*Myriophyllum triphyllum* and *Potamogeton ochreatus*). Turf species (*Lilaeopsis novae-zelandiae* and *Samolus repens*) showed an association with harder substrates and deeper sulphide depth as these species are found at the shallow lagoon margins.



### Ecological threshold response curves

Boosted regression tree models indicated good to fair amounts of deviance explained by the environmental parameters to species abundances. The best fit was obtained for *Ulva intestinalis* (D2 = 0.71), followed by *Ruppia polycarpa* (D2 = 0.45), *Ruppia megacarpa* (D2 = 0.38), and finally *Cladophora* spp. (D2 = 0.34).

The relative influence of environmental parameters was highly variable upon vegetation type (Figure 16). The strongest parameters for *Ruppia megacarpa* were sulphide depth, water depth and dissolved oxygen concentration. Depth and sulphide depth were also the most influencing factors for *R. polycarpa*, although the strengths of parameters was relatively uniform (Figure 16). For macroalgae, *Cladophora* spp. responded primarily to turbidity, whereas *Ulva intestinalis* also had quite a homogenous response, yet sulphide depth, temperature and salinity were the major drivers.

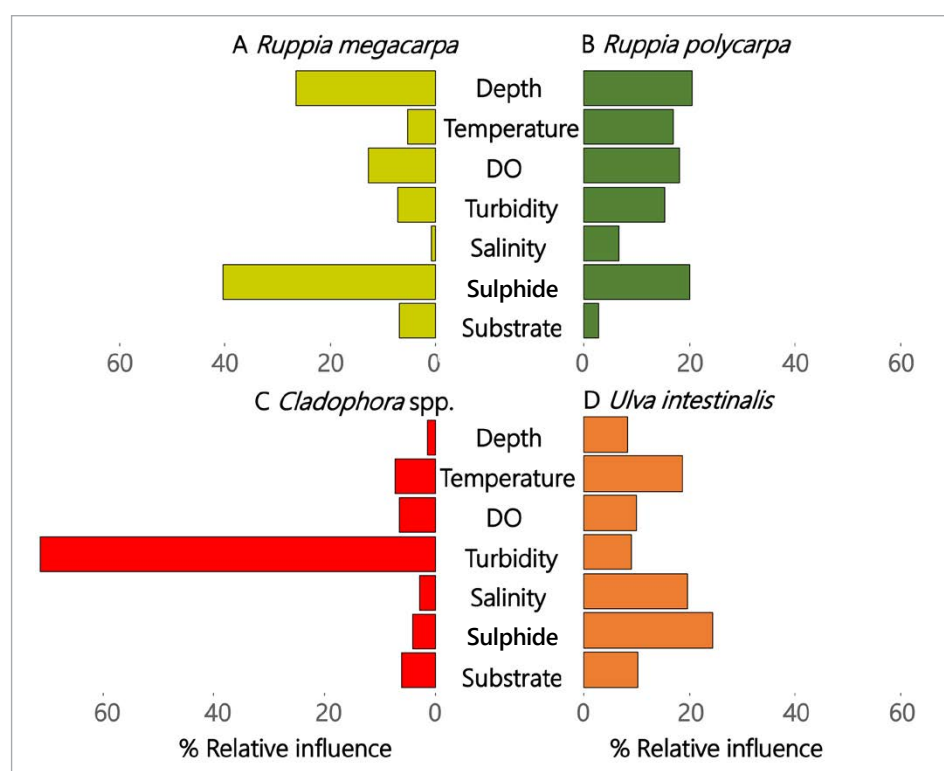


Figure 16: Barplot showing the relative influences of the seven environmental variables as indicated by boosted regression tree models for: (A) *Ruppia megacarpa*, (B), *Ruppia polycarpa*, (C) *Cladophora* spp. and (D) *Ulva intestinalis*.

Partial dependence plots (Figure 17) illustrated threshold response curves of the vegetation types to each environmental factor. The four vegetation types all showed reduced presence in the shallow depth range (depth <0.5 m). *Ruppia polycarpa* showed a shallower depth preference centred on 1 m depth, compared to *Ruppia megacarpa* that preferred a depth of 1.5 m or more. *Ulva intestinalis* had a shallower depth preference than *Cladophora* spp., which had a wide depth range that was possibly associated with each of the *Ruppia* depth ranges.

For surface temperature, *R. megacarpa* was associated with cooler temperatures (in keeping with deeper depths) and decreased over 15°C to 18°C. *R. polycarpa* exhibiting wider temperature tolerances to between 16°C and 20°C. In contrast, both macroalgae groups showed strong associations with higher temperatures of 19°C or more.

*R. megacarpa* and *Cladophora* spp. showed an association with higher surface DO. Similarly, for *R. polycarpa* most of the relationship showed a similar pattern. However, there was no convincing relationship between *U. intestinalis* abundances and DO (not shown).

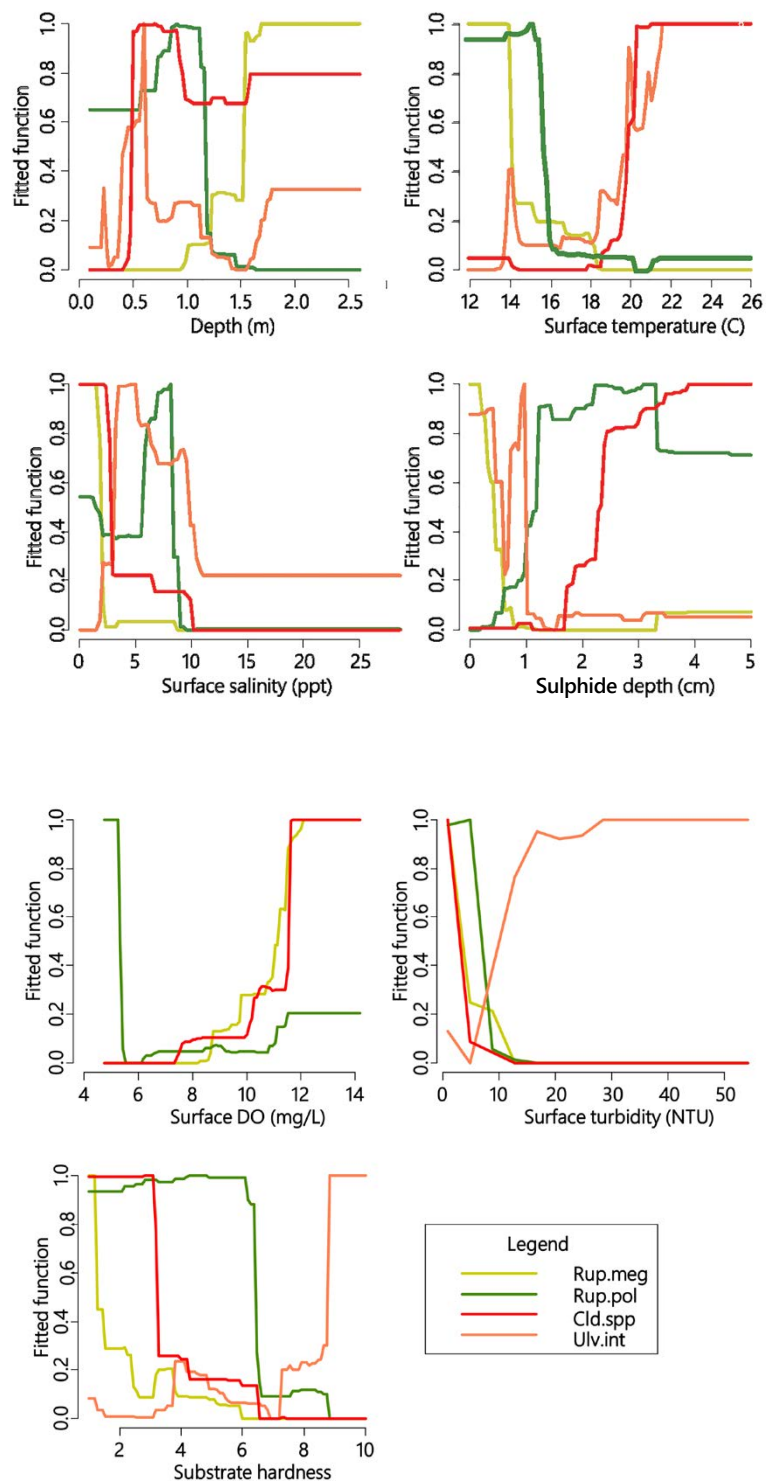


Figure 17: Partial dependence plots from four boosted regression tree models of the abundance of four species (as indicated in the legend) and for seven parameters. For species codes see Table 1, page 20. The relationship between *Ulv.int* and DO was not convincing and, therefore, was removed.





*Ruppia* and *Cladophora* spp. had a narrow tolerance range for turbidity at up to 5 to 10 NTU, but *Ulva intestinalis* showed no such limitation. This relationship may be confounded by eutrophication processes and wind-wave action both influencing turbidity in Waituna Lagoon.

*R. megacarpa* showed the lowest salinity tolerance of <4 PSU, while *R. polycarpa* appeared to tolerate 8-9 PSU. *Cladophora* spp. showed lower salinity preferences (<2.5 PSU) than the *U. intestinalis* (2.5 to 10 PSU).

*R. megacarpa* was associated with a blackened sulphide layer positioned close to sediment surface, but *R. polycarpa* mainly occurred where the sulphide layer was deeper than c. 1 cm. *U. intestinalis* had intermediate preferences between the *Ruppia* species. *Cladophora* spp. associated with deeper sulphide layers than the other vegetation groups.

*R. megacarpa* was associated with soft substrates, while *R. polycarpa* was associated with a wide range of substrate hardness. *Cladophora* spp. were more usually associated with muddy substrates, while *U. intestinalis* was predominantly associated with gravel substrates.

## Discussion

The ordination and ecological threshold response curves show that submerged plants and macroalgae types responded differently to physico-chemical conditions in Waituna Lagoon in summer, which are in turn strongly influenced by lagoon mouth status.

Elsewhere environmental factors of salinity, turbidity and water depth were found to drive changes in the abundance of *R. megacarpa* in an Australian estuary (Carruthers et al. 1999). Amongst the most important variables for *Ruppia* species in Waituna Lagoon were water depth, sulphide depth, and DO. However, different strengths of these factors were evident between the species and they were separated in ordination space. Previously, *R. megacarpa* and *R. polycarpa* have been described as occupying different ecological niches (Nicol 2005).

Within the response curves of all vegetation types to water depth, it appeared there was a non-vegetated 'varial' zone at the lagoon edge. This may correspond with low water level under an open lagoon regime where vegetation is desiccated. These impacts may be added to by wind-driven turbulence at the sediment surface (Herbst et al. 2013), dislodgement of shoots, as well as similar affects associated with wind-seiche in the lagoon.

Carruthers et al. (1999) found that abundance of *R. megacarpa* was positively correlated with water depth in Wilson Inlet, Western Australia. Moreover, *R. megacarpa* in Australia is generally considered a perennial associated with permanent waters (Brock 1985). This is in keeping with the ordination for Waituna Lagoon where *R. megacarpa* showed a preference for deeper water.

The difference in depth response curves between the two macroalgal groups may be due to the partially benthic nature of *U. intestinalis* (initially attached to stones), compared to the benthic, epiphytic and floating mat growths of filamentous green macroalgae dominated by *Cladophora* spp.

Sulphide depth had the strongest relative influence on response curves for three out of the four species, particularly for *R. megacarpa* which was associated with a near-surface sulphide layer. Although this study did not measure sulphide levels directly, it is a known phyto-toxin that causes root/rhizome mortality in vascular submerged plants (Azzoni et al. 2001). Plant species are known to have differential sensitivity to sulphide (Pedersen and Kristensen 2015) and some species are even dependent on some reduced sulphur in



sediments (Pulich 1989). *Ruppia* species are known to be able to detoxify sulphides in shallow sediments by releasing oxygen by the roots (Azzoni et al. 2001, Gennaro et al. 2004). The association of *R. megacarpa* with a shallow sulfide layer suggests that it has not modified the sediment redox profile via root oxygen release and is instead tolerant of sulphide, or that these sediment conditions confer another advantage for this species.

*U. intestinalis* was also associated with a shallow sulphide layer, and elsewhere high concentration of sulphates were among the most important factors affecting biometric parameters of *Ulva* blooms (Rybak 2018). Macroalgal mats can also create sediment anoxia (Han and Liu 2014), although this did not hold true for *Cladophora* spp., which was associated with deeper sulphide layers than the other vegetation groups.

DO appeared in the ordination to structure *R. megacarpa* and was one of the stronger relative influences in the response curves for both *Ruppia* species and macroalgae dominated by *Cladophora* spp. However, one interpretation is that DO production from vegetation during the monitoring day has led to this association, rather than it being a preference.

Temperature was one of the stronger relative influences for both macroalgae groups which increased at higher temperature. Temperature was also identified as a major variable for describing the habitat of an *Ulva* species in freshwater systems (Rybak and Gąbka, 2018). *Cladophora* dominated growths also responded to higher temperature in keeping with reported high optima for growth of this species of 28-30°C (Lester et al. 1988).

Elsewhere, salinity gradient was the most important factor determining the spatial distribution of submerged vegetation within a lagoon, with other variables of importance being sediment organic content and grain size, as well as TSS (Burgos-Leon et al. 2013). Surprisingly, salinity did not have a strong relative influence on response curves of *Ruppia* species in Waituna Lagoon. It is apparent that salinity levels in Waituna Lagoon did not approach the critical thresholds reported for these *Ruppia* species. For instance, salinity tolerances of 45 PSU were reported for *R. polycarpa* survival in an Australian ICOLL (Nichol 2005) and for survival of adult *R. megacarpa* under experimental conditions (Sim et al. 2006). Moreover, *R. megacarpa* is known to persist in hypersaline systems in Australia (Brock 1982) and in New Zealand (Wells et al. 2010). However, we note these records may be from systems where salinity increases gradually through evaporation (e.g., Kim, 2013). Elsewhere conductivity had a negative relationship to the abundance of *R. megacarpa* despite being less than reported salinity tolerance limits (Carruthers et al. 1999). Where there has been a lack of *Ruppia* tolerance to salinities within reported tolerance ranges, there have been rapid salinity fluctuations in the field, linked to plants inability to osmoregulate under dynamic salinity conditions (Strazisar et al. 2015)

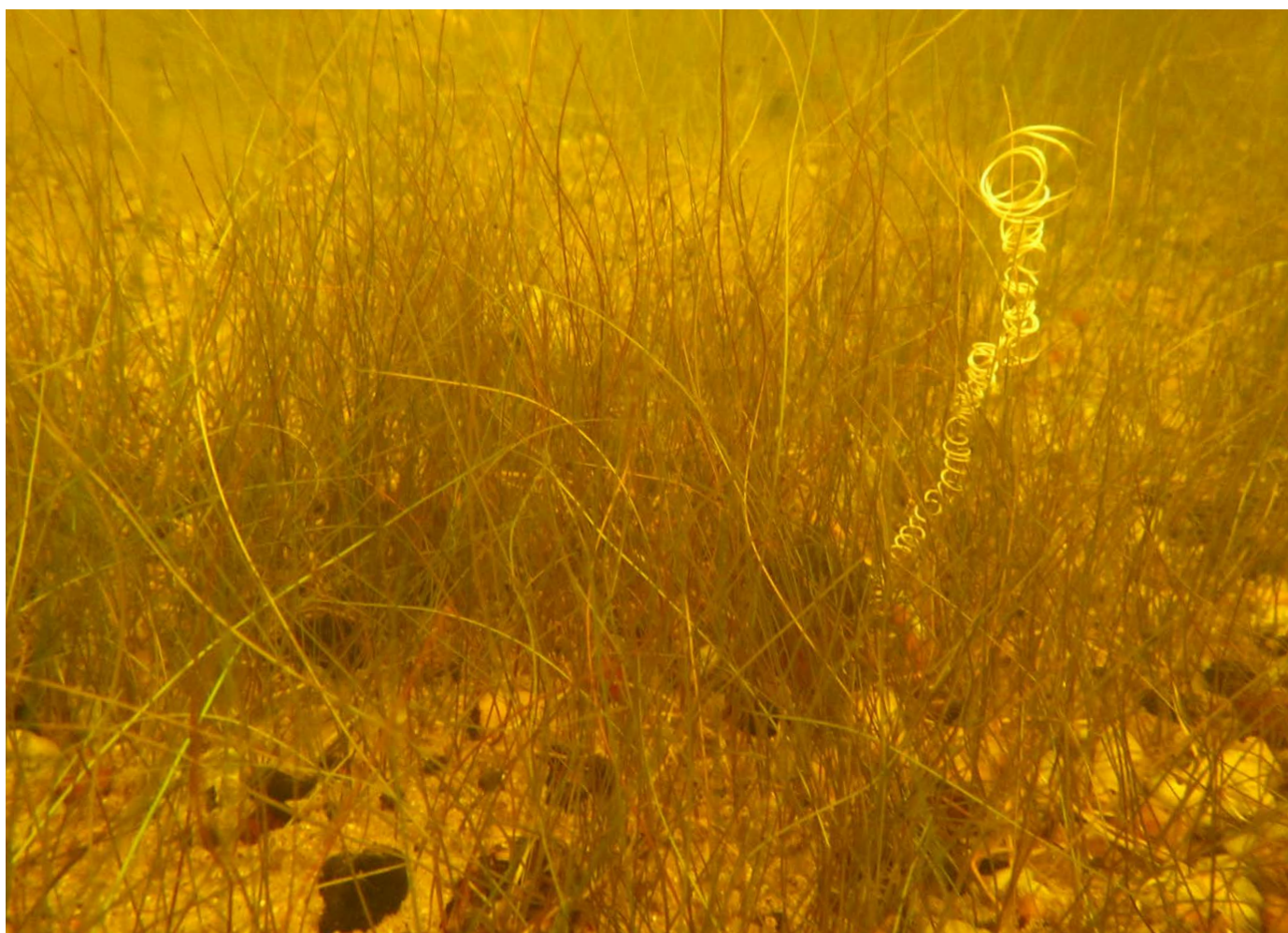
*Ulva intestinalis* was associated with higher salinity in the ordination for Waituna Lagoon and salinity had a strong relative influence on response curves for this species. This finding is in agreement with increases in *Ulva* spp. with sustained salinity increases in a South African ICOLL (Riddin and Adams 2010) and reflects the estuarine habitat of this species. By contrast, green filamentous algae dominated by *Cladophora* spp. were more strongly associated with freshwater conditions in both the ordination and in the response curves.

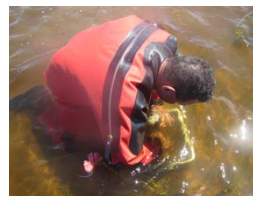


The species with the strongest substrate preference in response curves for Waituna Lagoon was *U. intestinalis*, probably on account of its need for hard substrate for spore settlement.

The ordination and response curve results link *R. megacarpa* with soft sediment, but elsewhere *R. megacarpa* has been associated with coarse-grain sediment (Carruthers and Walker 1999). Submerged vascular plants are known to increase local sedimentation and reduce grain size, particularly where they have a high biomass (Bos et al. 2007). Therefore, it is plausible that *R. megacarpa* has modified the substrate at the sites where it is recorded rather than soft substrate being a habitat preference.

Increased turbidity can confer depth constraint for plants through reducing light availability in deeper water. Both *Ruppia* species showed a preference for low turbidity levels, but conversely *R. megacarpa* preferred deeper water. *R. megacarpa* biomass was inversely proportional to turbidity in an Australian estuary (Carruthers et al. 1999). Interestingly, there was a very strong relative influence of turbidity on response curves for *Cladophora* spp. Possibly the incompatibility of macroalgal mats with wave-disturbed shorelines accounts for this negative association.





## Informing Future Lagoon Management

It is likely that a complex of conditions associated with mouth status interact to determine vegetation composition and abundance. For instance, in this analysis salinity was not the most influential driver for vegetation.

There appeared to be a shallow 'varial' zone of low abundance for the *Ruppia* species and main macroalgae in the response curves. This is likely to represent disturbance, which at low water level could be related to wave action, or currents generated by tidal fluctuations and to desiccation. *Ruppia* beds are located at an elevation of 0.25 to 1 m above mean sea level (Hamilton et al. 2012), therefore, an open lagoon would lead to higher desiccation risk. However, a seiche operating in the lake could lead to local shoreline desiccation at any lagoon level. These findings support lake openings in winter when evaporative rates are lower and freshwater inflows might keep lagoon levels higher. Interestingly, modelled climate scenarios suggested that higher *Ruppia* biomass was associated with years with higher than average rainfall by affecting water levels in the lagoon (Hamilton et al. 2012).

*Ruppia polycarpa* had wide tolerances based on the response curves. Consequently, we would expect this species to persist under a range of conditions and to be able to recover from minor perturbations. Widespread presence of *R. polycarpa* is valid as an indicator of lagoon health, however, additional indicators aimed at ability to recover from major perturbation (e.g., an extended opening or drought) should focus on reproductive success and likely maintenance of a seedbank. We recommend an ecological target based on flowering success is developed for the lagoon.



Ordination and response curves suggest that *R. megacarpa* is associated with deeper lagoon areas than *R. polycarpa*. Monitoring results also highlight the association of this species with higher vegetation development and, therefore, influence on the environment. Anecdotal observations of reduced wave action in areas of *R. megacarpa* beds would confirm this species is a strong 'autogenic engineer'. If significant vegetation influence is sought for the lagoon then maintaining deeper lagoon levels should lead to greater development of *R. megacarpa*. Development of an ecological target for *R. megacarpa* development would capture trends for lagoon health.

Analysis of SoE data suggests that nutrients were elevated in winter. Ordination and response curves show macroalgae development increases with increasing temperatures. Therefore, these findings support the strategy to open the lagoon in winter to flush seasonally elevated nutrients before temperatures increase to avoid macroalgae blooms.

Triggers and conditions in the 2017 Resource Consent seek to ecologically optimise the timing and conditions of lagoon openings and already address some of the aspects identified in this analysis.

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