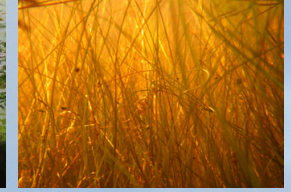




Department of
Conservation
Te Papa Atawhai



NIWA
Taihoro Nukurangi



Technical Report on Vegetation Status in Waituna Lagoon: 2009–2020



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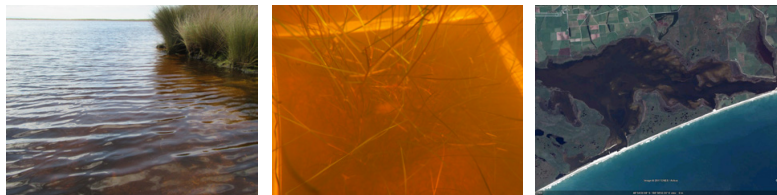
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Introduction

This technical report accompanies the summary report on vegetation status in Waituna Lagoon in 2020 (de Winton 2020). We review the lagoon conditions over the period of vegetation monitoring from 2009 to 2020 and update current lagoon vegetation status according to findings in 2020.

As background to the summary report, this technical report describes water level, mouth status and duration (Section 1). The report also summarises recent lagoon conditions based on monitoring of indicators of water quality carried out by Environment Southland (Section 2). We provide descriptions of monitoring methods and presents summaries of data and analyses (Sections 3, 4 and 5). We briefly conclude what the findings mean for lagoon management.



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 water level. Lagoon closure was estimated from timing of subsequent, sustained increases in level. The total time period for openings was calculated, the lagoon mouth status was confirmed and the duration of that status before each vegetation monitoring event was calculated as months (one month is 30 days).

Results

Prior to the 2020 survey of vegetation, Waituna Lagoon had been open to the sea for 4.1 months (Figure 1). This opening lasted the entire spring/summer period (September to February) that constitutes the main growth period for *Ruppia*. Therefore, the target of three months of closed conditions prior to vegetation monitoring (Lagoon Technical Group 2013) was not achieved in 2020. Similar spring/summer openings were recorded prior to surveys undertaken in 2011, 2013, 2014 and 2017 (Figures 1 and 2). The lagoon has previously recorded open conditions for 3.9 to 6.2 months prior to monitoring occasions on three of these occasions (Figure 1, negative axis).

Over winter 2019, two lagoon openings were made. The initial opening in mid-June lasted 47 days before closing, but rapidly rising water level necessitated a second opening in mid-October 2019 which then persisted (Figure 2). As previously, tidal exchanges and sea level following lagoon opening resulted in oscillations in water level around a lagoon height that was approximately 0.5 m less than normal level (Figure 2).

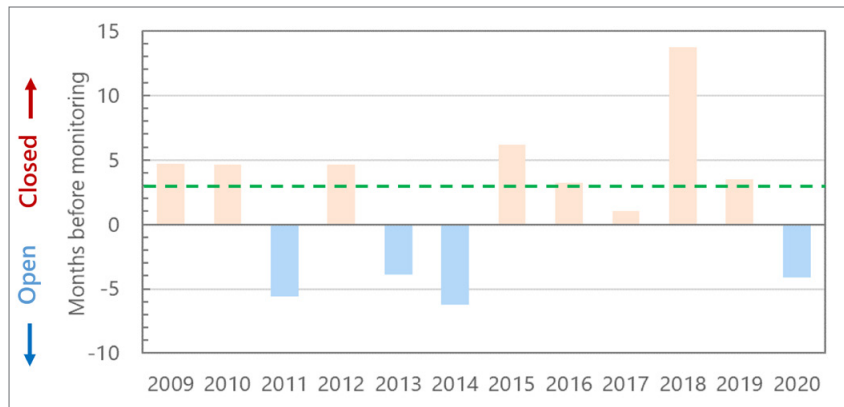


Figure 1: Diverging bar plot showing the number of months for which Waituna Lagoon was open or closed prior to monitoring (as indicated by the y axis). The dotted line indicates the ecological target of three months of lagoon closure before monitoring.

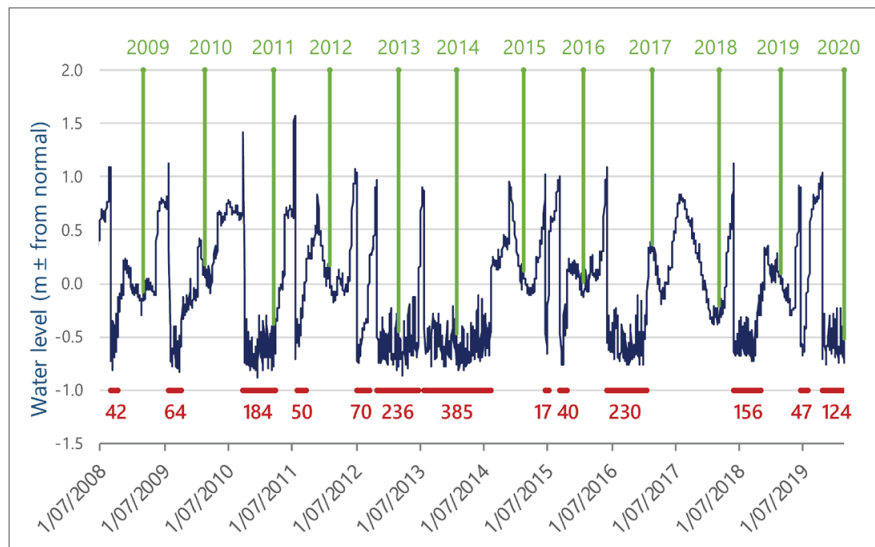


Figure 2: Plot showing the continuous water level time series for Waituna Lagoon, measured at Waghorns Road. Periods of lagoon opening are indicated by horizontal red lines. The number of days during which the lagoon was open correspond to the red numbers. Finally, the annual summer vegetation monitoring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

Discussion

The geomorphologic classification for Waituna Lagoon (Hume et al. 2016) suggests the natural lagoon state would have been closed to the sea, with short-lived openings. Coarse substrate comprising the coastal barrier permits drainage to the sea by percolation, which would reduce the water pressure leading to natural breaches. The barrier would breach naturally when sufficient pressure builds from high water levels in the lagoon, and/or severe storm waves overtop the barrier. Closure of the barrier is driven by the high energy wave-dominated action on a mixed sand/gravel coastline. Therefore, the natural occurrence of sustained openings would be of decadal–century time scales. In contrast, artificial openings are annual and have been sustained for up to a year at times (i.e., 2013–2014).

Current consent conditions for artificial openings of Waituna Lagoon (Resource Consent 20146407-01) have favoured winter openings since 2017. This means there is more opportunity for closure before the main spring/summer growth period for *Ruppia*. A short duration opening was seen in June–July 2019, but a later October opening had not closed between then and the 2020 monitoring event.



2. Temporal Physico-chemical Conditions

Methods

Water quality monitoring data for Waituna Lagoon was obtained from Environment Southland from 2009 to 2020. Data from the central lagoon sampling site was used to indicate changes in conditions over time to simplify temporal patterns. Seven parameters were plotted between 2009 and 2020:

1. Chlorophyll-*a* (Chl-*a*, mg L⁻¹)
2. Salinity (PSU)
3. Total Nitrogen (TN, g m⁻³)
4. Total Phosphorus (TP, g m⁻³)
5. Total Suspended Solids (TSS, g m⁻³)
6. Turbidity (NTU)
7. Temperature (°C)

Where water quality parameters were reported below detection limits, we plotted a value equal to half that detection limit. Timing and duration of lagoon openings is indicated in relation to water quality parameters.

Results

Large variations in monitored parameters occurred seasonally and also with mouth status and interannually (Figure 3). The lagoon opening in October 2019 was associated with a sharp increase in salinity with records of over 25 PSU in January and February 2020, as was measured during some other opening events (Figure 3a). Temperature followed a seasonal pattern, increasing over the duration of the most recent opening.

The October 2019 opening was also followed by a drop in nutrient levels (TP and TN), and chlorophyll-*a* concentration, as is apparent after some other openings (Figure 3b). Measures of suspended solids and turbidity also decreased over the most recent opening event (Figure 3c).



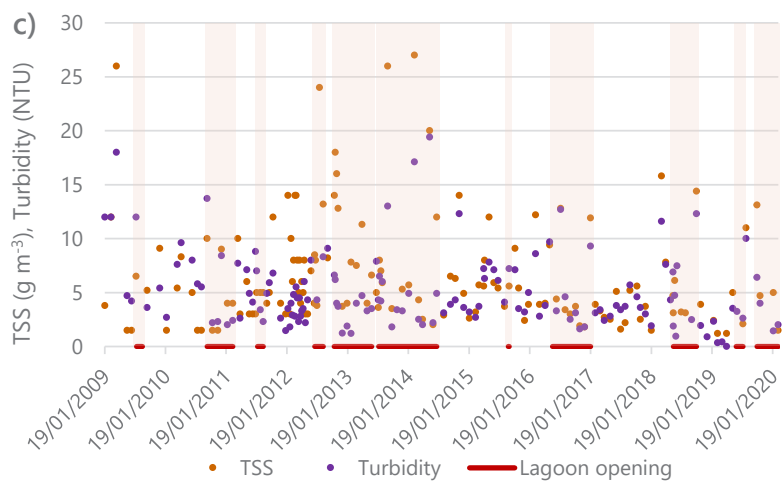
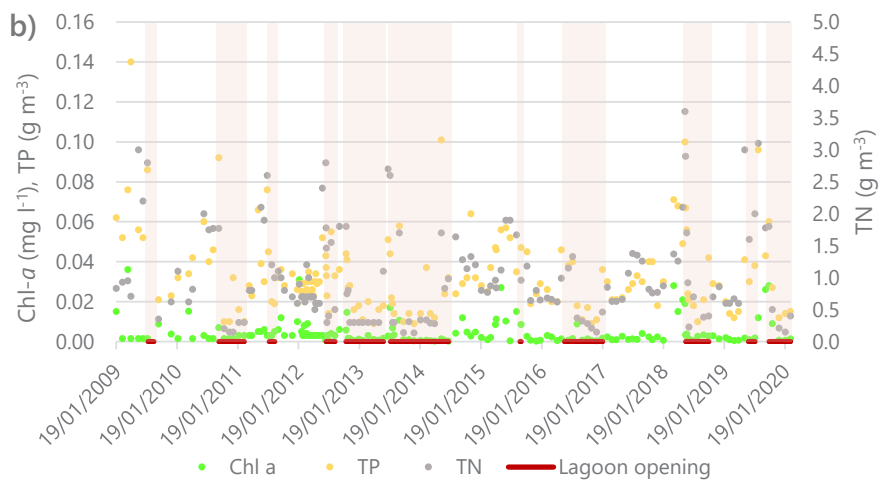
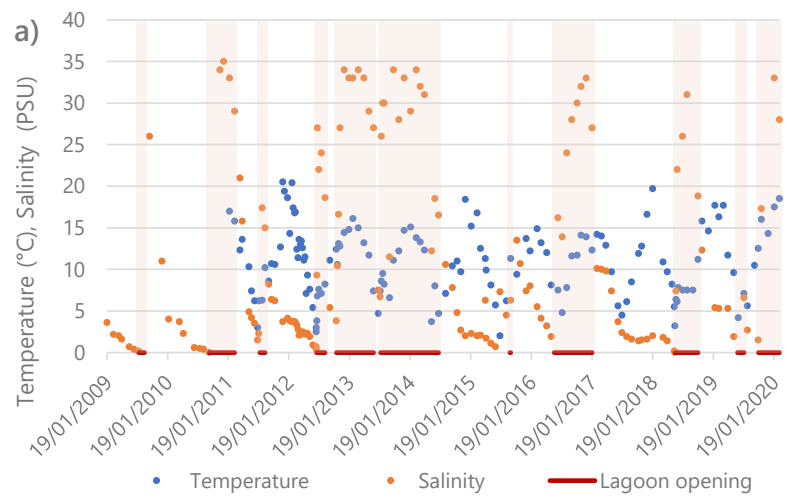


Figure 3: a) Time plots of temperature and salinity, b) Chlorophyll-*a* (Chl-*a*), total phosphorus (TP), and total nitrogen (TN), and c) Total suspended solids (TSS) and turbidity at the lagoon centre sampled over 2009 to 2020.

Discussion

Status and timing of lagoon openings have previously been shown to be a major driver of physico-chemical conditions in the lagoon (Schallenberg and Tyrell 2006, Schallenberg et al. 2020, Hodson 2017, de Winton and Mouton 2018). In turn, physico-chemical conditions have been linked to the spatio-temporal patterns of the vegetation in Waituna Lagoon (Robertson and Funnell 2012, Lagoon Technical Group 2013, de Winton and Mouton 2018).

The 2020 vegetation monitoring follows a long (>3 month) lagoon opening, which is generally associated with increased salinity, decreased temperature (relative to season), decreased nutrients and total suspended solids (de Winton and Mouton 2018). However, strong seasonal signals in temperature, nutrients and suspended solid concentrations are also noted (de Winton and Mouton 2018).

In the following section (Section 3) we describe the physico-chemical conditions at the time of monitoring in 2020 and compare with previous annual monitoring over a range of lagoon mouth status conditions.



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 from 2014 onwards).

At each monitoring site, measurements were made from 2009 to 2020 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:

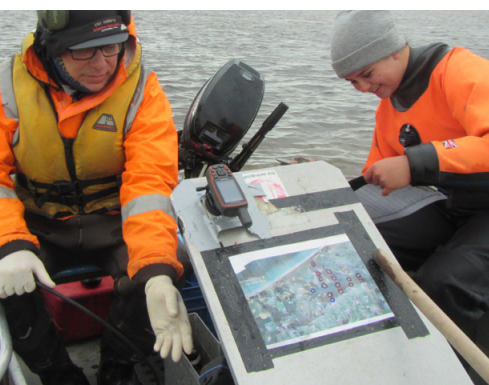
- Temperature (°C).
- Dissolved oxygen (DO, mg l⁻¹).
- Salinity (PSU).
- Turbidity (NTU).

Black disk, DO and turbidity were measured from 2011 to 2020.

The surface and bottom water quality measurements were previously found to be highly correlated (Spearman $r > 0.9$, de Winton and Mouton 2018), we therefore employed average values for each parameter. Four sites were exposed at the time of monitoring in 2020 due to low tide conditions. For those sites, water quality measurements were made at closely adjacent inundated areas. The data was then illustrated using box plots for each year (each annual monitoring event).



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



Results

At the time of the 2020 monitoring, Waituna Lagoon had been open to the sea for 4.1 months. Salinity levels were high (average 25.2 PSU), similar to levels recorded in 2011 and 2014 (Figure 5) when the lagoon had been open for over five months. Salinity was also higher in 2017 (average 10.3 PSU), when the lagoon had only been closed for a month before monitoring. Highest salinity levels in 2020 (≥ 30 PSU) were recorded at the ten closest sites to the barrier opening.

Although monitoring sites tended to be shallow in 2020 (average 0.52 m), water depth was slightly deeper than was recorded during other monitoring events when the lagoon was open to the sea in 2011, 2013 and 2014 (Figure 5). This may be related to the tidal cycles during the respective monitoring. Four sites that were dry during 2020 monitoring had been recently inundated. Relatively shallow depths recorded during monitoring in 2018 (Figure 5), despite the lagoon being closed, were due to drought conditions in Southland. The remainder of monitoring events were deeper under closed lagoon conditions (Figure 5).

Water temperature in 2020 (average 18.6°C) was amongst the four highest recorded years (including 2013, 2015 and 2017) (Figure 5). Water temperatures have been noticeably higher since 2012.

DO averaged 8.9 mg l⁻¹ at monitoring sites in 2020 (Figure 6). Although nine sites recorded DO below 85% saturated, a value usually considered well oxygenated, no sites fell below 5 mg l⁻¹, which is generally considered a threshold necessary for healthy aquatic life.

In 2020, turbidity averaged 14.8 NTU and was relatively high in comparison with previous years of annual survey (Figure 6). As in some previous years, results were skewed by upper outliers (>30 NTU), which in 2020 comprised wind-wave disturbance of sediment in shallow water at the northern shorelines. Correspondingly, water clarity measured by black disk in 2020 was lower than previous years (averaging 0.8 m) except for 2011, 2012 and 2018 (Figure 6).



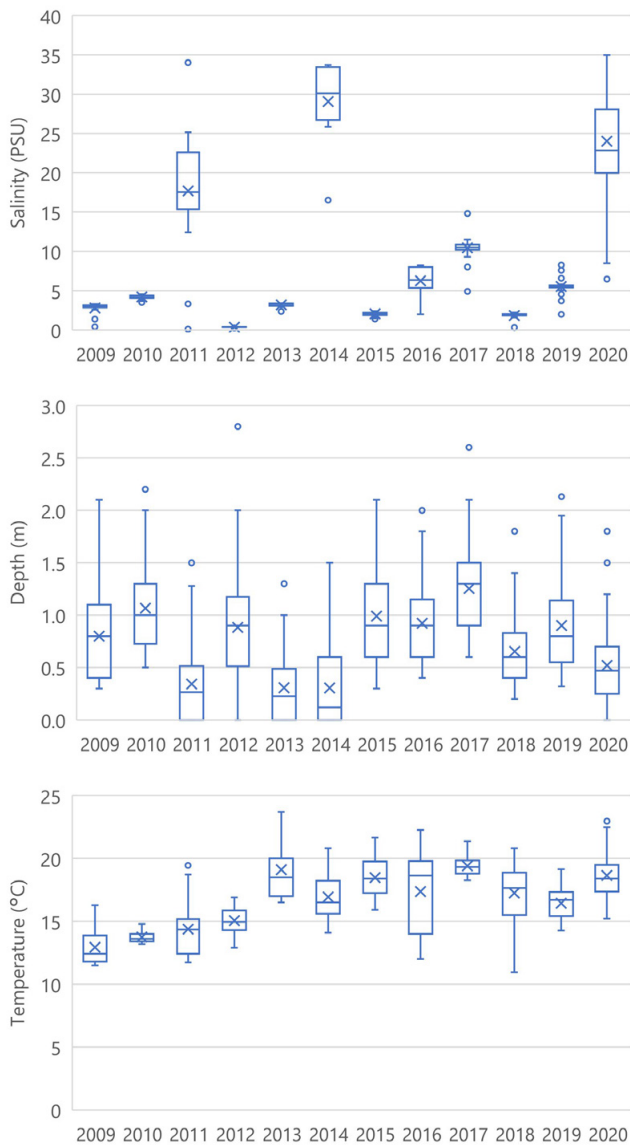


Figure 5: Box and whisker plots of salinity (top), depth (middle) and temperature (bottom) over all monitoring years. (n= 48 or 47).

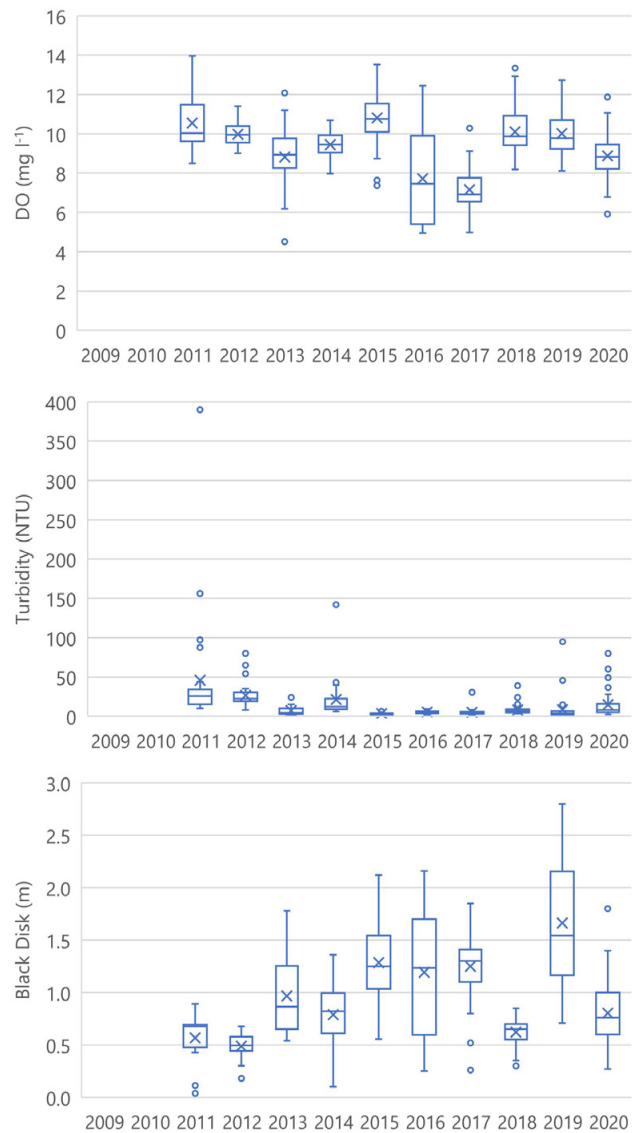
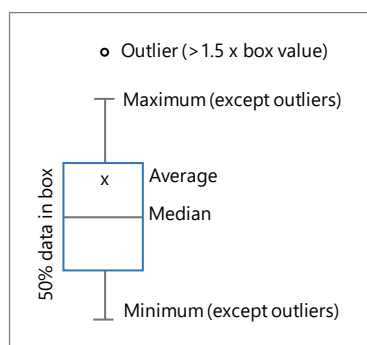


Figure 6: Box and whisker plots of DO (top), turbidity (middle) and black disk (bottom) at the monitoring sites (n= 48 or 47), from 2011 to 2020.



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

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 status of the lagoon mouth. In keeping with the extended open lagoon status, higher salinity and turbidity were recorded in 2020. Although water clarity was lower, reflecting both turbidity and water colour, the shallower depths at sampling sites would tend to compensate for a lower transmission of light through lagoon waters and permit *Ruppia* growth.

Temperature was relatively high at the time of the 2020 monitoring, differing from the previous findings that an open lagoon generally results in cooler conditions. However, we note that physico-chemical measurements taken during the vegetation monitoring event are ‘snap-shot’ only and likely to be influenced by short-term meteorological events.



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 observed.

Results

In 2020, sampled sites had a smaller proportion of the softer substrates (categories 1–3) than sampling in 2019 (Figure 7). Similar levels of soft substrate to 2020 were recorded in 2012–2014, 2016 and 2018. By contrast, harder substrates in 2020 (categories 7 to 10, Figure 7) were also reduced relative to 2019, and more similar to 2009–2012 and 2018. Therefore, most sites were characterised in the mid-range of sediment hardness (categories 4–6) in 2020 (Figure 7).

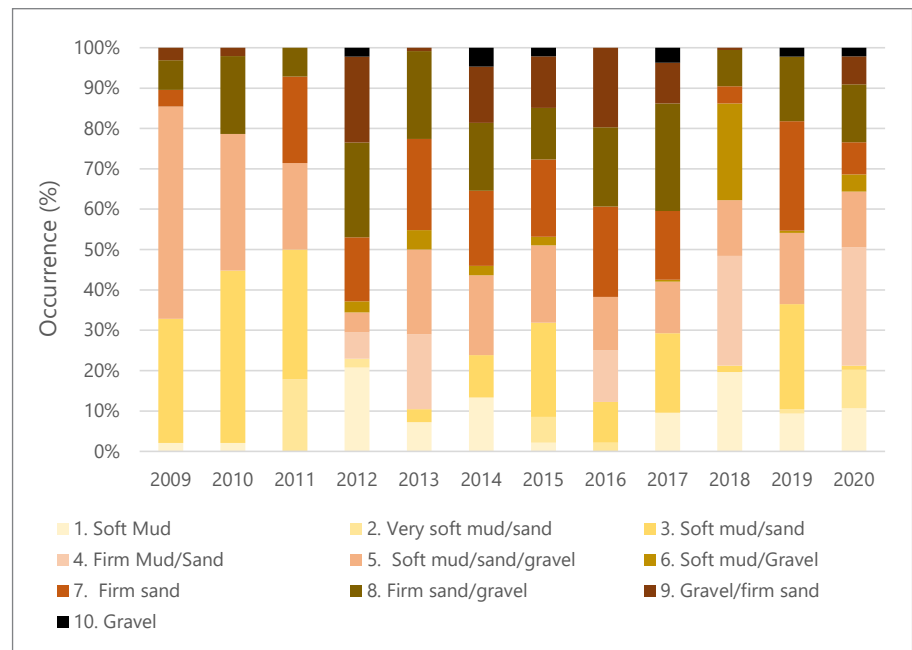


Figure 7: Bar plot illustrating the composition in substrate type (% occurrence), recorded during each of the annual monitoring surveys. Substrate types are numbered from softer to harder.



In 2020, a blackened layer in the top 2 cm of sediment comprised 29% of samples, similar to the proportion recorded in 2019 (Figure 8). But the proportion of sites without a blackened layer had increased from 2019 to over 50% of sediment samples in 2020. This proportion was similar to 2014 after an extended lagoon opening. However, we noted that a number of samples ($n = 20$) in 2020 had a mottled appearance where a distinct blackened layer could not be discerned.

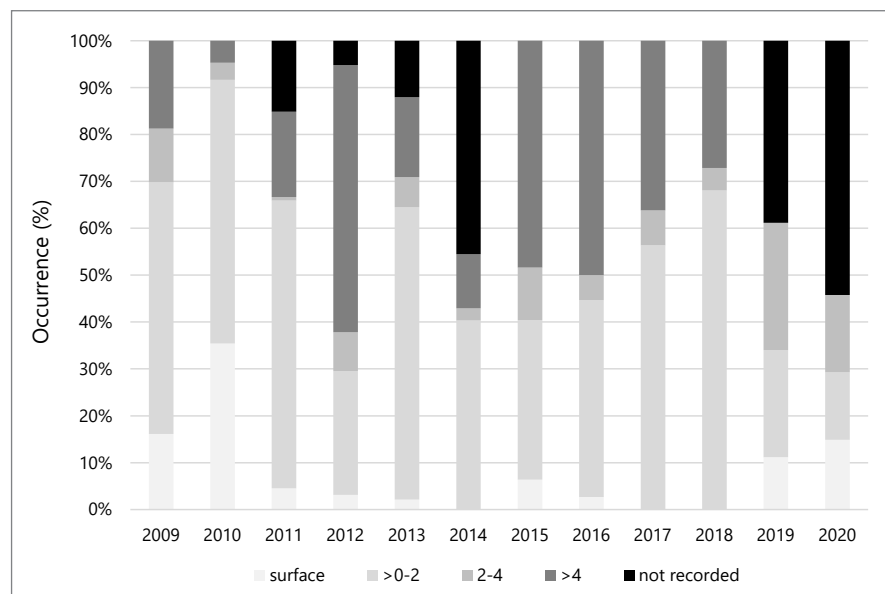


Figure 8: Substrate depth to a blackened layer shown as occurrence (% records) for five incremental depth categories.

Discussion

The reduction in soft substrates in 2020, together with increases in the proportion of substrates in the mid-range for hardness, suggests some re-distribution of substrates by tidal currents generated under the open lagoon status. Harder substrates are associated with a better ecological condition (Stevens and Robertson 2007).

A shallow sediment depth to the blackened layer may indicate where substrates have a shallower anoxic layer. Anoxia can be associated with nutrient release from sediments or the build-up of substances that are toxic for plant growth. In 2020, the depth to the blackened layer had generally moved deeper within the sediment profile, indicating an improvement (sediment surface is less anoxic and 'healthier'), but there were some ambiguous samples that could not be categorised.



5. Vegetation Development

Methods

At each site (Figure 4), 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 range¹ 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 range² 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 onwards, macrophyte observations were also 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

Three sites recorded no vegetation (plants or macroalgae) from the hoe samples in 2020 (Figure 9) with none of these sites comprising 'dry' sites. The most widespread submerged plants in hoe samples were *Ruppia polycarpa* (28 sites) and *R. megacarpa* (10 sites) with an overall 29 sites recording *Ruppia* species (Figure 9). Other submerged plants that were occasionally encountered in samples were *Myriophyllum triphyllum* (5 sites) and the charophyte *Lamprothamnium macropogon* (2 sites). The amphibious turf plant *Lilaeopsis novae-zelandiae* was recorded at 1 site.

Macroalgae were widely recorded in 2020, with *Ulva intestinalis* present at 28 sites and filamentous green algae, dominated by *Cladophora* and/or *Ulothrix* species, at 24 sites (Figure 9). A 'brown' filamentous alga that was encountered on vegetation and substrate (7 sites) was identified as *Audouinella* species (Figure 9).

Hoe samples in 2020 showed relatively few sites without any vegetation (Figure 9) compared with previous years when the lagoon had similarly been open for c. 4–6 months in 2011 (29 sites), 2013 (26 sites) and 2014 (32 sites). However, these previous years had a much lower contribution by macroalgae.

The overall contribution of *Ruppia* species to vegetation composition in 2020 was similar to 2015–2017 and to 2019. *Ruppia* species were 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* species in Figure 9). The freshwater macrophyte *Myriophyllum triphyllum* was substantially reduced in occurrence in 2020, compared to the consecutive closed lagoon mouth years of 2018 and 2019 (Figure 9).

The charophyte *Lamprothamnium macropogon* also strongly decreased in occurrence in 2020, and has been more conspicuous in contributing to vegetation in the years the lagoon was closed including 2012, 2015–2016, 2018–19 (Figure 9).

¹ 1 = 1–5%, 2 = 5–10%, 3 = 10–20%, 4 = 20–50%, 5 = 50–80%, 6 = 80–100%

² <5 cm, 5–15 cm, 15–30 cm, 30–50 cm, 50–80 cm, 80–100 cm





Macroalgae in 2020, comprised *Ulva intestinalis* at a similar occurrence to that recorded in 2019 (Figure 9), but filamentous green macroalgae comprised a greater proportion of recorded vegetation. These two categories of macroalgae became more prominent components of vegetation in 2015-2017 and 2019 (Figure 9). The diatom *Bachelotia antillarum* has not been prominent since 2014 (Figure 9).

In contrast to recent years, *Bachelotia* was noted as widespread in 2009 and 2010 (Robertson and Stevens 2009, Stevens and Robertson 2010). However, as this diatom forms a loose covering on the sediment surface, the hoe did not sample this alga successfully and results were not able to be adequately quantified or included in the results plotted in Figure 9.

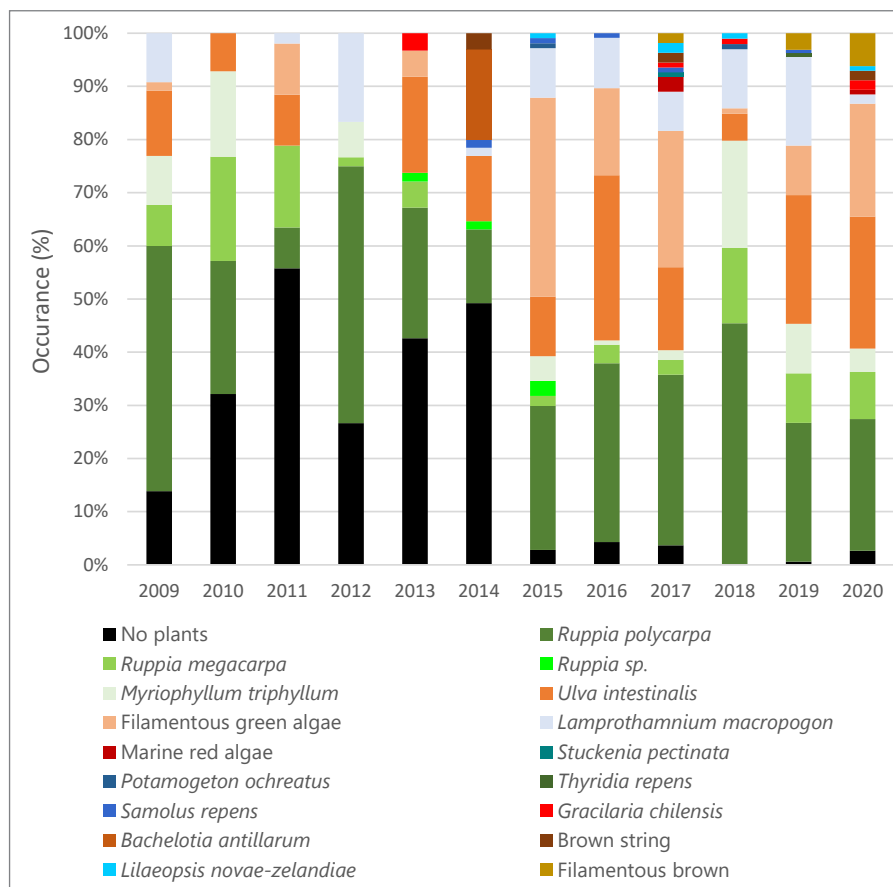


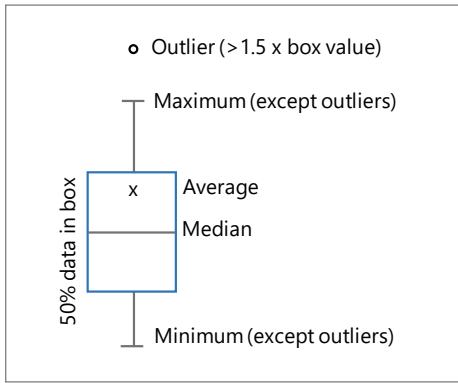
Figure 9: Vegetation composition shown as relative frequency of occurrence (sites recorded) for species or vegetation groups.



Ruppia abundance

The cover of *Ruppia* measured by the hoe method in 2020 averaged 5.4%. This cover value represented a large decrease from 2019 and was similar to low covers (<10%) measured in 2011–2014 and 2017 (Figure 10). High *Ruppia* covers ($\geq 80\%$) were recorded in hoe samples at two sites in 2020 that were dominated by *R. megacarpa*. Similarly, 58% of hoe samples that recorded high covers ($\geq 80\%$) over all surveys comprised *R. megacarpa*, which was disproportionate to the species occurrence. The height of *Ruppia* in 2020 (average 8.8 cm) was also reduced relative to 2019 (Figure 10) and was similar to levels recorded in 2011 and 2013 (≤ 10 cm). The two upper outlier sites for height measured in 2020 comprised *R. megacarpa*. *R. megacarpa* has also comprised 61% of all height records ≥ 50 cm that have been measured to date through all monitoring years.

‘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*. Biomass index averaged 199 in 2020, which was similar to 2012–2014 but higher than 2011 (Figure 10). In contrast, average



biomass index exceeded 1000 in 2015, 2016, 2018 and 2019 (Figure 10). As common in years before, high outliers for biomass index in 2020 represent sites with *R. megacarpa*.

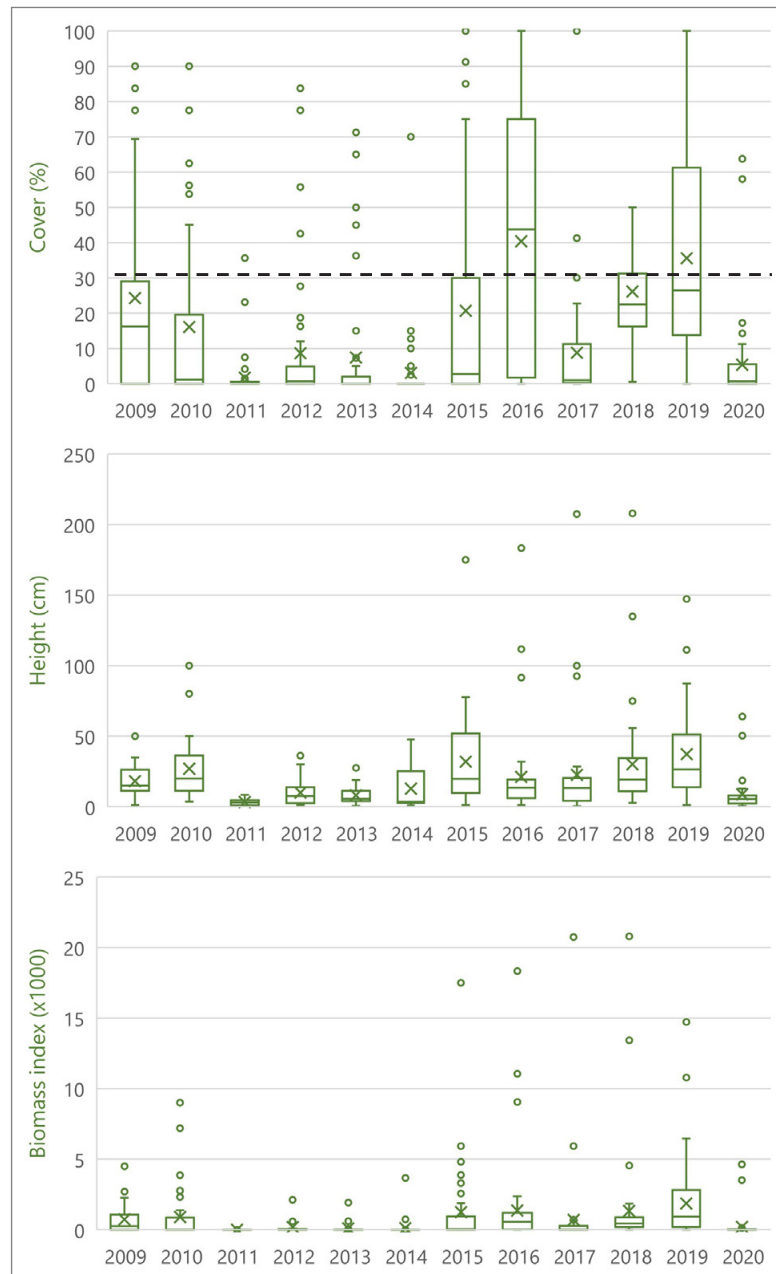


Figure 10: Box and whisker plots of *Ruppia* cover (top), height (middle) and derived biomass index (bottom) over monitoring years. Average of measurements at monitoring sites (n= 48 or 47). Dotted line represents the lagoon-wide target for *Ruppia* cover of 30% identified by the Lagoon Technical Group (2013).



Macrophyte observations by divers at sites (all years since 2013) have correlated with the results obtained from the hoe method (Figure 11). Observations tended to give higher average covers than the hoe method, particularly where hoe covers were low (Figure 11). This may reflect the patchy nature of *Ruppia* clumps. However, in 2020, the average lagoon-wide cover of 5.7% for *Ruppia* using the diver observations was very similar to the lagoon-wide estimate of 5.4% cover using the hoe method. Diver observations similarly showed low average *Ruppia* covers of less than 12% for monitoring years 2013, 2014 and 2017 with covers exceeding 30% for the years 2015, 2016, 2018 and 2019. Average height of *Ruppia* in 2020 from diver observations was 17.5 cm, higher than those based on hoe samples (Figure 11), but this result likely reflects the greater chance of encountering tall plants in the wider assessed area.

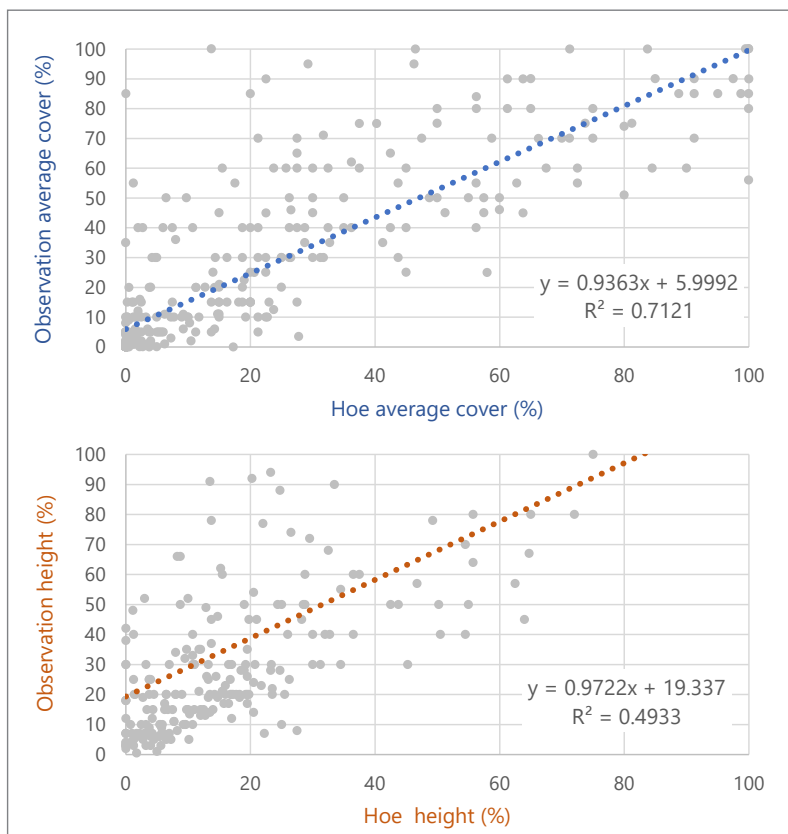


Figure 11: Relationship between *Ruppia* cover (top) and height (bottom) estimated from hoe samples and diver observations within a 10 m diameter area at each site.

Ruppia life-stage

Reproductive status of *Ruppia* in 2020 was amongst the lowest recorded (Figure 12). Just 6% of observations were for flowering or post flowering samples from the hoe method, similar to 2011, 2013, 2014 and 2017 (Figure 12) when the lagoon had also been open for most of the *Ruppia* spring-summer growth period.

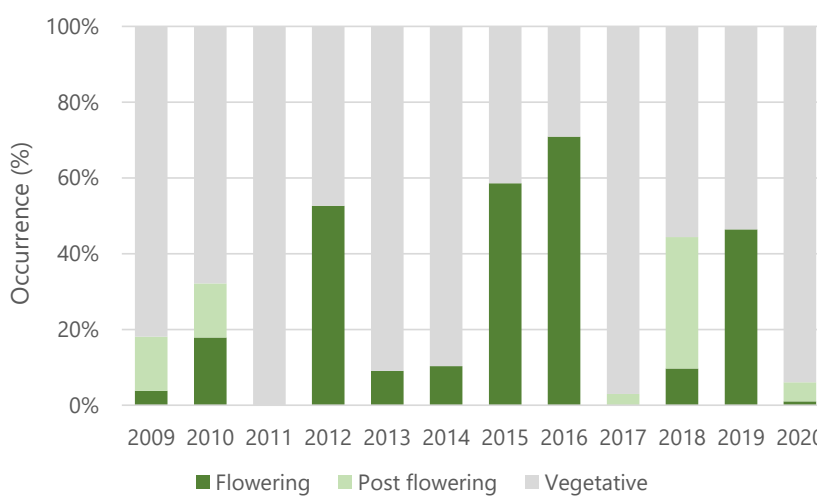
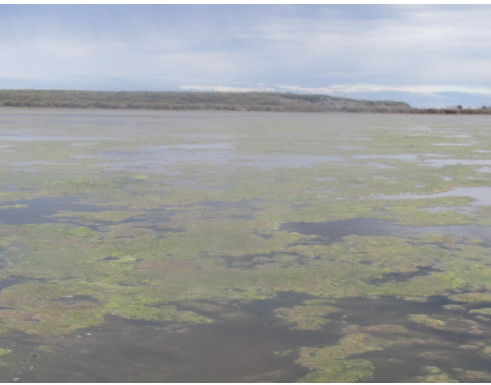
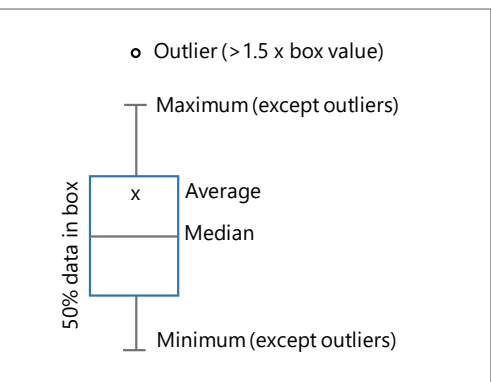


Figure 12: Life-stage category of *Ruppia* species across monitoring years as a proportion of records.



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



Macroalgal cover

In 2020, macroalgae cover based on the hoe method at an average of 25% had decreased relative to the high levels of 66% cover recorded in 2019 (Figure 13). Similar covers to 2020 were seen in 2015–2017 (average 20–45%) that were substantially higher on average than 2009–2014 and 2018 (Figure 13). In 2020, most macroalgae were benthic or epiphytic. In some years, particularly 2015–17, macroalgae exceeded 100% cover at some sites due to combinations of benthic, epiphytic, and floating growths (Figure 13).

There remains a weak correlation (all monitoring years) between macroalgal cover estimate by hoe samples and overall covers observed by divers (Figure 14). It appears that macroalgal covers are likely to be under-estimated 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 hoe method that constitutes benthic sampling.

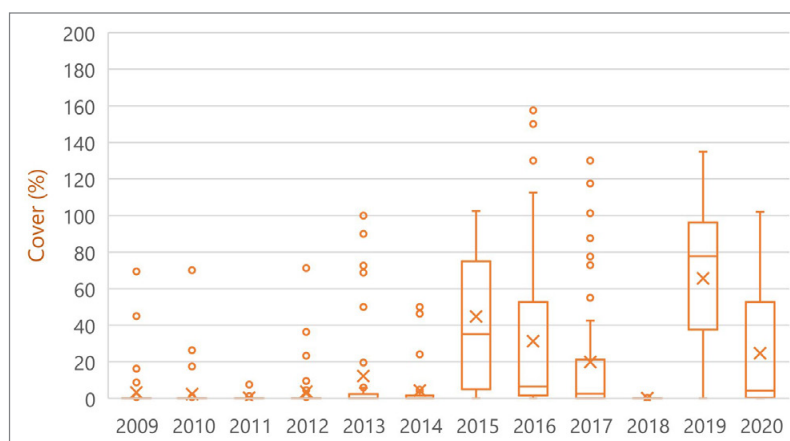


Figure 13: Box and whisker plots of macroalgae cover over monitoring years as an average of measurements at monitoring sites (n= 48 or 47).

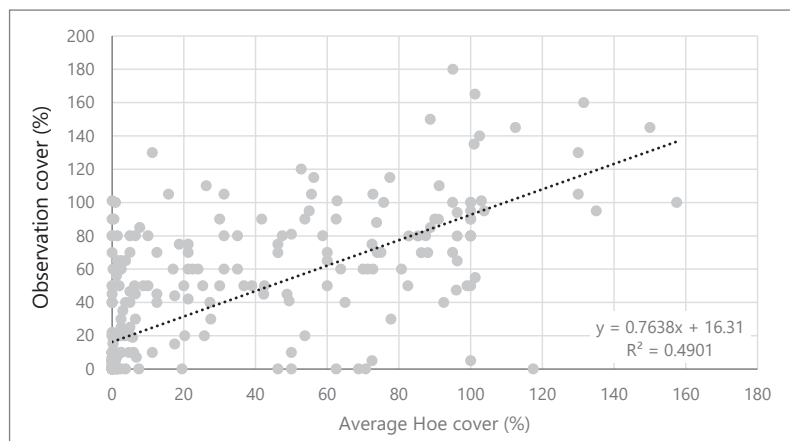
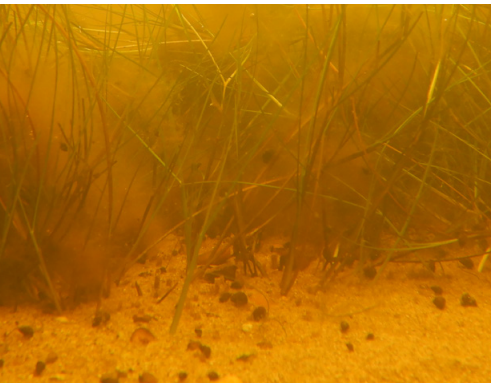


Figure 14: Relationship between macro-algal percentage cover estimated from hoe samples and diver observations within a 10 m diameter area at each site.

Discussion

Ruppia abundance was low in 2020 following the open lagoon status over the entire spring-summer growth season. This monitoring event followed the 2018 and 2019 monitoring years when the lagoon was consecutively closed over spring-summer, and *Ruppia* had a much higher abundance. While cover and height of *Ruppia* was reduced in 2020, plants remained distributed widely across the lagoon (68% of sites) and, therefore, are well positioned



to recover during a subsequent favourable period for growth. *R. megacarpa*, a beneficial ‘ecosystem engineer’ (an organism capable of modifying a habitat), also remained a significant component of vegetation composition in 2020.

Ruppia reproduction seems to be strongly influenced by lagoon mouth status. It may be that the generally small stature of *Ruppia* plants in 2020, particularly *R. polycarpa*, influenced plants ability to flower and fruit. Elsewhere flower abundance has been linked to above ground biomass of *Ruppia* species (Santamaría et al. 1995) and flowers must reach the water surface for successful pollination. Poor fertility under high saline conditions has been noted for *Ruppia* species in Australian wetlands (Sim et al. 2006).

Although 2020 was a poor year for *Ruppia* reproduction, the previous successful fruiting observed in summer 2018 and 2019 will have replenished the seed bank of these species. The seed bank for these species is perennial (de Winton and Mouton 2018), but while viability extends for greater than one year, longevity is unknown. Seed banks would provide an important means of vegetation recovery following severe perturbation. In 2020, we documented large reservoirs of *R. megacarpa* seed on the sediment surface at Hansen’s Bay (Site 4.5), with several having germinated despite the high salinity (22 PSU) recorded at this site (Figure 15).



Figure 15: *Ruppia megacarpa* seed collected from one hoe sample in Hansen’s Bay showing several germinated seed with white roots.



Macroalgae also remained widespread in 2020, although their abundance had decreased from the high levels seen in 2019. Lagoon mouth status does not appear to be such a strong driver of macroalgae development as it is for *Ruppia* development. Meteorological events and their influence on nutrient loading and temperature conditions in the lagoon are likely to play more of a major role in macroalgal development. We did not find a strong negative impact by macroalgae on *Ruppia* abundance that is apparent in monitoring results to date. The data may not be at sufficient scale (especially temporally) to relate *Ruppia* abundance to macroalgae cover. However, we also note that shading impacts by macroalgae on submerged plants is likely to be greater where surface floating mats develop under warm still conditions.

Continuation of additional diver observations is useful where patchy vegetation distribution means small scale hoe sampling may undersample vegetation. Benthic hoe sampling has some limitations for the retrieval of macroalgae in particular.



Informing Future Lagoon Management and Research

The 2020 vegetation monitoring follows a long (>3 month) opening, which for Waituna Lagoon is associated with increased salinity, decreased temperature (relative to season), decreased nutrients and total suspended solids (de Winton and Mouton 2018). Strong seasonal signals in temperature, nutrients and suspended solid concentrations are also apparent, but lagoon mouth status is a major signal within the summer period (de Winton and Mouton 2018).

Water quality measurements taken by NIWA during the annual *Ruppia* survey in 2020 are a 'snapshot' of open lagoon conditions, but data show strong spatial variability across the lagoon (e.g., salinity gradients). Data from Environment Southland for the period leading up to the 2020 monitoring event appear to confirm the effect of lagoon opening in 2019 in flushing nutrients and reducing chlorophyll-*a*, suspended solids and turbidity. However, monthly data is a relatively coarse temporal scale for determining lagoon conditions in response to openings or meteorological events. Finer scale water quality data, especially on a temporal scale would be useful for interpreting vegetation responses, particularly for the macroalgae which seem to be more temporally dynamic than *Ruppia*.

Annual vegetation monitoring results at Waituna Lagoon over 12 years strongly suggest that an open lagoon status over the spring to summer plant growth season (e.g., for at least three months prior to summer monitoring) leads to lower *Ruppia* development and reproductive success than conditions associated with a closed lagoon or short duration winter openings. Therefore, the main drivers of summer annual status of *Ruppia* appear related to physicochemical conditions created by lagoon mouth status.



Previous analysis (de Winton and Mouton 2018) suggested that a complex of conditions including salinity, depth, desiccation and disturbance levels, water clarity and temperature are likely to interact in determining vegetation composition and abundance in Waituna Lagoon. In 2020, high salinities were associated with a greater reduction in the occurrence of the freshwater plant milfoil than by *Ruppia*. This is in keeping with other findings that increasing salinity favours persistence of *Ruppia* species over milfoil species (Hillmann and La Peyre 2019).

Salinity might influence *Ruppia* abundance, however there is little evidence for a mass *Ruppia* die-off in the lagoon when exposed to greater salinity. It has been noted that *Ruppia* can persist at high salinities, even hypersaline conditions. High salinity is known to reduce *Ruppia* growth rates (Gerbeaux 1989), which in turn seems to reduce reproductive success. What is unknown for Waituna Lagoon is the relative levels of physical disturbance and loss of *Ruppia* under open versus closed conditions. Increased losses under open lagoon conditions may result from desiccation, breakage and uprooting of plants from waves and tidal currents and increased waterfowl grazing access under low water level conditions. The net effect of these increased losses, together with lower growth and reproductive rates probably drives *Ruppia* population dynamics in the lagoon.

A widespread occurrence of *R. polycarpa* and sustained presence of *R. megacarpa* recorded in February 2020 mean Waituna Lagoon vegetation is likely to recover rapidly if suitable closed conditions are restored during the next main growing season. Sediment seed banks will provide a means of recolonisation should a sustained lagoon opening occur and further reduce *Ruppia* occurrence. However, this vegetation recovery would be much slower than vegetative recolonisation.

Macroalgae remained abundant in 2020 despite the open lagoon status. Relatively high temperatures were recorded at the time of the 2020 survey and are known to promote algal growth. Openings are thought to flush nutrients that build up under closed conditions and would otherwise drive high summer macroalgae levels. It may be that macroalgae abundance built up in 2019 has been able to persist under open lagoon conditions or that the macroalgae were not nutrient limited. For instance, macroalgal mats that accumulate on sediment surfaces in shallow environments may be sustained by the release of porewater nutrients into overlying waters (McGlathery et al. 1997).

Issues around the accurate sampling of macroalgae are apparent and measurements from annual monitoring are likely to have underestimated macroalgae presence. In future, we recommend that surface growths of macroalgae are separately recorded (presence at sample sites and mat thickness) as these growths have the greatest shading potential for macrophytes. We also note that macroalgae populations can respond over much shorter timeframes than rooted macrophytes (e.g., with certain meteorological conditions). This suggests that the annual monitoring alone is inadequate for describing macroalgae dynamics and other, higher frequency monitoring would be useful.

It appears that the recent lagoon opening also had a beneficial influence on sediment condition in 2020, which was better than recent monitoring results for a closed lagoon, with an apparent redistribution/flushing/processing of fine sediments and deepening of sulphide horizons.

Vegetation monitoring results from 2020 for Waituna Lagoon, when added to the time series, show further evidence to support short, winter openings as a means of protecting a widespread *Ruppia* vegetation and the ecological benefits that submerged plants provide. In contrast, lagoon openings that have persisted through more than one *Ruppia* growth season have resulted in much reduced vegetation presence and a slow recovery.

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