Weed management and flooding in the Pūharakekenui/Styx River

Final report

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Prepared by:

Fleur Matheson
Daniel Clements
Sanjay Wadhwa
Elizabeth Graham
Iñigo Zabarte-Maeztu
Aleki Taumoepeau
Susie Elcock
Matt Moss

For any information regarding this report please contact:

Dr Fleur Matheson
Biogeochemist
Aquatic Plants
+64 7 856 1776
fleur.matheson@niwa.co.nz

National Institute of Water & Atmospheric Research Ltd
PO Box 11115
Hamilton 3251

Phone +64 7 856 7026

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	Reviewed by:	Mary de Winton Rob Davies-Colley
	Formatting checked by:	Carli Nolan
	Approved for release by:	Neale Hudson

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Executive summary

The Pūharakekenui/Styx River is a lowland, spring-fed system that runs along the northern boundary of Christchurch. Since 1990, the water level in the lower river has been managed to prevent flooding of surrounding low-lying land by harvesting aquatic weeds. However, anecdotal observations suggest that this management method has been less effective in recent years, and that the weed growth is more vigorous. The Comprehensive Stormwater Network Discharge Consent (CSNDC) requires the Christchurch City Council (CCC) to investigate the impediment of drainage by excessive aquatic weed growth under Schedule 4 item (r). This includes assessment of various options for managing river channel weed to mitigate these effects.

NIWA was contracted by the CCC to develop a scope for the investigation in September 2020, and to undertake this investigation from November 2020 to July 2022. The investigation included the following components:

- A delimitation of nuisance weed distributions using a combination of broad-scale and fine-scale assessment methods.
- A review of weed harvesting data and information provided by CCC, and by CityCare, who carry out the harvesting operation.
- A trial to evaluate the efficacy of the aquatic herbicide diquat to effectively manage the weed nuisance in the river.
- An evaluation of riparian shade, its relationship to weed growth in the river, and the potential for additional riparian plantings to add further shade and reduce weed growth.
- An exploratory analysis of available monitoring data to identify potential hydrometric, water quality and climate factors responsible for increased weed growth, supplemented with a targeted assessment of sediment nutrients.

The focus area for the investigation was the c. 14.6 km reach of river mainstem from Redwood Springs downstream to the tide gates.

Weed delimitation was carried out using broad-scale hydroacoustic mapping and fine-scale channel cross-section surveys. Mapping and surveys were repeated seasonally, in spring, summer and autumn.

A trial to evaluate diquat efficacy was conducted from March to June 2022. Diquat was applied to a 200 m reach of the river, just upstream of Radcliffe Road, in March. Aquatic plants were surveyed, and water quality monitored before and after diquat application, in this reach, an upstream control reach, and a third reach downstream.

Our review of weed harvesting was limited to the data provided. Water level reduction data associated with weed harvesting for the period 2000 to 2017 were analysed to identify spatial and temporal trends. Harvesting records for 2020-21 were tabulated, and reductions of water level arising from harvesting events were determined using available hydrometric data.

Riparian shade was measured during the cross-section surveys and local effects on weed abundance were evaluated. Riparian shade and vegetation height were modelled along the river length, using LiDAR data and ArcGIS software tools. The potential for new riparian plantings to increase shade along the river length was evaluated by determining the extra vegetation height that would be required at regular intervals along the river's length to achieve shading in excess of 70%. Previous New Zealand research by the authors has indicated that this level of shading will constrain weed growth substantially.

Water level, flow, water temperature, nutrient, total suspended solids, salinity, conductivity, air temperature, solar radiation, and sunshine hours data for the period 2007 to 2017 were compiled for an exploratory analysis. These data were supplemented with sediment nutrient data acquired during a targeted sampling campaign during 2021. A trend detection analysis was performed on the long-term data. The potential for sediment and water nutrients to maintain aquatic plant growth was assessed.

Canadian pondweed (*Elodea canadensis*, or elodea) and the curlyleaf pondweed (*Potamogeton crispus*), both introduced species, were the dominant aquatic weeds identified in our surveys. The native submerged plants, stonewort (*Nitella* sp. aff. *cristata*), blunt pondweed (*Potamogeton ochreatus*) and common water milfoil (*Myriophyllum propinquum*) were also present but were much less abundant. Aquatic vegetation was most dense in autumn, occupying c. 30% of the entire wetted channel volume between Redwood Springs and the tide gates, and up to 61% of cross-sectional area. A shallow, flow-constricted area (c. 1.3 km length) was observed while undertaking hydroacoustic surveys in the lower river, presumably caused by sediment accrual.

Water levels were reduced appreciably by weed harvesting activities, by about 16-27 cm in summer 2021, and by up to 73 cm based on past records. Analysis of data from 2000 to 2017 indicated a likely increasing trend in efficacy of harvesting to reduce water levels, and thus flooding-risk, in the mid-lower river, contradicting anecdotal reports of recent decreasing efficacy. The analysis also showed that harvesting was more effective when carried out later in the growing season (January to May) than earlier (September to December), suggesting greater operational efficacy when weed is denser.

Application of diquat had a small to negligible (i.e. statistically insignificant) effect on weed abundance, with total weed clogginess reduced by c. 7% in the application reach compared to the upstream and downstream reaches. Elevated river flows at the time of treatment, reducing contact time, and increased concentrations of particulate and dissolved constituents that can adsorb diquat, may have contributed to the low diquat treatment efficacy. Diquat was not detected in water samples from the river collected in the 2-to-24 hour period after application.

Aquatic plant abundance in the river was inversely correlated with shade. Measured shade ranged from 0 to 64%. Modelling using LiDAR consistently over-estimated shade, probably because the modelling cannot account for light transmission through the canopy. Our analysis indicated that in locations along the riverbanks with low stature vegetation, new plantings of dense vegetation that add 10 m of extra riparian vegetation height, on one or across both banks, would reduce weed accrual along much of the river length.

We identified increasing trends since 2007 in water levels, total and baseflows, air and water temperatures, and solar radiation, and decreasing trends in water nutrients. Sediment and water nutrient content in 2020-21 were low to adequate for aquatic plant growth and are considered unlikely to contribute to excessive weed growth and flooding potential.

Key findings of the investigation are:

- Increasing water levels, and therefore risk of flooding in the lower Pūharakekenui/Styx River are most likely due to a trend of increasing baseflow discharge, potentially exacerbated by increased flow impedance caused by aquatic weed and sediment accrual.
- The aquatic weed, *Egeria densa* (or egeria), is present in the nearby Avon River where the weed harvester also operates. There is a substantial risk that this plant may be transferred to the Pūharakekenui/Styx River as fragments on the harvest, and this requires immediate action to reduce the potential for establishment of the species.
- Riparian planting to increase shade of the river is expected to reduce aquatic weed growth generally and increase conveyance of the channel, mitigating flood risk even when higher discharges occur.
- Diquat application resulted in a small reduction in weed nuisance, but trial conditions were not ideal, and further evaluation may reveal greater efficacy.

Recommendations arising from this investigation are as follows:

- Identify and implement an appropriate method for decontaminating the harvester to minimise the risk of egeria introduction from the Avon River by this means.
- Undertake annual surveillance to maximise the potential to detect any early incursion of a more aggressive aquatic weed species, such as egeria, *Lagarosiphon major* (lagarosiphon), or *Ceratophyllum demersum* (hornwort), and develop a response plan to ensure preparedness should an incursion occur.
- Train or provide guidance to CCC and CityCare staff doing river monitoring and maintenance, to enable them to identify these high-risk weed species, and provide signage to inform the general public.
- Investigate the removal of sediment accrued downstream of the Spencerville Road Bridge.
- Undertake further trials under stable river flow conditions to better evaluate the efficacy of diquat herbicide as an additional option for management of weed nuisance in the river.
- Increase riparian plantings to a height equal to the wetted width in more open sections of the river, ideally on both banks, ensuring plantings create a dense screen.
- Ensure regular calibration of the stage/discharge relationship at Radcliffe Road as this can be strongly affected by weed growth.

1 Introduction

The Pūharakekenui/Styx River is a lowland, spring-fed system that runs along the northern boundary of Christchurch. The river originates in the Christchurch suburb of Harewood and travels in a north-east direction for approximately 23 km before flowing into Te Riu o Te Aika Kawa/Brooklands Lagoon and the Waimakariri River, at its mouth. The river has a relatively small (70 km^2) catchment and meanders through reserve, pasture, horticultural areas and residential developments. It has two main tributaries, the Kāpūtahi/Kaputone Stream and Smacks Creek which are spring-fed but also receive stormwater inflows. As the river is predominantly spring-fed, baseflow conditions tend to dominate. Water levels in the river are strongly affected by growth and accumulation of weed in the river channel, particularly in summer to autumn. The lower reach is tidal, but with saline intrusion and tidal amplitude reduced by tide gates.

Since 1990, the water level in the lower river has been managed by weed harvesting to prevent flooding of surrounding low-lying land. Harvesting has been necessary to manage water level, even at base flow. However, observations suggest that this management method has been less effective in recent years, and weed growth has been more vigorous, despite an unprecedented dieback of weed in 2018 due to an unknown cause. The most recent river surveys (van den Ende and Partridge 2008) indicate that the submerged exotic macrophytes, *Elodea canadensis* (hereafter elodea) and *Potamogeton crispus* (hereafter curlyleaf pondweed) are the main nuisance species.

The Comprehensive Stormwater Network Discharge Consent (CSNDC) from Environment Canterbury requires Christchurch City Council (CCC) to investigate the impediment of drainage by excessive aquatic weed growth under Schedule 4 item (r). The investigation programme is also required to assess the degree to which various options for river channel weed management are likely to mitigate potential for flooding under a range of river flow scenarios.

NIWA was contracted by CCC to develop a scope for the investigation which was completed in September 2020. A field visit undertaken as part of scoping found no evidence for incursion of more aggressive weed species, such as *Egeria densa* (hereafter egeria), *Lagarosiphon major* (lagarosiphon) or *Ceratophyllum demersum* (hornwort). These introduced species have the potential to increase weed nuisance in any waterway within which they establish. The scope was peer reviewed by three CCC-appointed technical experts, and the final scoping document was submitted in October 2020.

Weed management options considered during the scope development included: mechanical harvesting, riparian shading, aquatic herbicides (diquat, endothall), nutrient limitation, grass carp and periodic saline intrusion. The use of grass carp and endothall herbicide were excluded from further consideration. Grass carp were excluded because their introduction was likely to devegetate and destabilise the river system, and they would be difficult to contain on site and remove once introduced. Endothall was not considered suitable because it does not control elodea and it requires a long contact time with weed to be effective, which is difficult to achieve in flowing water.

In November 2020 NIWA was contracted to deliver the following services during 2020/22:

1. Delimitation of weed nuisance.

   The full extent of the weed nuisance in the Pūharakekenui/Styx River has not recently been assessed and a contemporary delimitation was considered necessary to guide future weed management. The delimitation included the following steps:

(a) Hydroacoustic mapping, and ground-truthing observations, were carried out to show the spatial and temporal distribution of weed bed extent, where possible, in the mainstem of the Pūharakekenui/Styx River, downstream of Redwood Springs.

(b) A series of ten representative channel cross-sections along the river length were surveyed to show the proportion of watercolumn occupied (% clogginess) by different macrophyte species.

(c) Mapping was repeated once in spring, summer, and autumn to indicate seasonal shifts in weed species abundance and distribution[1]. It was carried out independently of the harvesting operation.

2. Review of weed harvesting.

In the current stormwater consent, a (presumably baseflow) water level that is at or above a Reduced Level[2] of 10.1 m at the Lower Styx Road monitoring gauge triggers the need for harvesting, which operation must then commence within 6 weeks.

A review of available and new data was recommended to identify opportunities for improved future management, with a focus on the 2020-21 season.

This component was dependent on receipt of data (GPS-tracking data and maps of the harvest operation) for the 2020/21 season from the CCC and CityCare (as operators of the harvester unit). Also requested were records of the weight of vegetation removed in harvested areas and estimates of proportional contribution by different weed species.

Unfortunately, the high-resolution data requested[3] for our review could not be provided. Consequently, only data about the harvesting operation that CityCare and CCC could supply was reviewed. This consisted of CityCare data on weed quantities removed by harvesting in different segments of the river in the 2020-21 season, and CCC data on water level reductions attributable to harvesting in past seasons.

3. Diquat efficacy trial.

This trial was intended to evaluate the efficacy of diquat herbicide for control of elodea and curlyleaf pondweed in the Pūharakekenui/Styx River, and to evaluate effects on non-target aquatic plants and river dissolved oxygen levels. To the authors' knowledge, a trial of this kind has not been carried out in the river previously. However, a trial was carried out in the nearby Avon River in 2001 in response to an incursion of egeria, which showed that it could effectively control this species (Wells and Sutherland 2001).

[1] It was also proposed that if specific coordinates for the 2007 (van de Ende and Partridge 2008) survey could be sourced, the earlier survey method should be replicated for a selection of representative sites, ideally aligned with channel cross-sections above to identify whether significant changes in macrophyte communities have occurred since 2007. However, these coordinates were not available. An assessment of changes in the river macrophyte community prior to 2000 was made by Taylor et al. (2000).

[2] The Reduced Level is the level above the Christchurch Drainage datum plane.

[3] using a GPS system (ideally incorporating real-time kinematic (RTK) positioning), to show the cutting lines, velocity of the craft, amount of time taken and areas of operation for each day during all harvesting operations.

A small-scale trial was carried out in autumn 2022. The trial included the following steps:

(a) Identifying a suitable reach for diquat application as well as a control reach (within 100 m upstream, not subject to treatment) and a third reach (within 1 km downstream). All reaches were c. 200 m in length without significant inflows.

(b) Surveying aquatic macrophytes in the reaches prior to diquat application (one to two days before) and twice thereafter (4 weeks and 2-3 months after application).

(c) Deploying loggers in the three reaches to continuously measure dissolved oxygen, water temperature and water level for the duration of the trial.

(d) Measuring diquat in water leaving the application reach at intervals of 2, 4, 6, 8, 12 and 24 hrs after application, and the same for the upstream control reach (to ensure no upstream movement of diquat due to tidal influences).

4. Riparian shade evaluation.

Shade from riparian vegetation has potential to reduce growth of aquatic weeds if it is sufficiently high, and there may be opportunities to increase riparian planting and shading of the Pūharakekenui/Styx River for this purpose. This option to improve aquatic weed control was therefore evaluated through a combination of field measurements and modelling.

The evaluation included the following:

(a) During the seasonal cross-section surveys, shade was measured across the channel on all 10 transects to determine present state.

(b) Using ArcGIS tools and LiDAR data, the amount of shade present in the river downstream of Redwood Springs was predicted, and riparian vegetation height was determined. The amount of additional shade which might be achieved with new plantings, to enhance weed control, was then determined.

(c) The seasonal cross-section measurements were used to validate the shade predictions.

5. Water quality, sediment, and climate effects.

   The perceived trend of increasing weed nuisance in the Pūharakekenui/Styx River system may be related to changes in water quality, sediment, and climate factors. This possibility was examined through an exploratory analysis of existing data, which included the following steps:

   (a) Available data on water levels, flows, salinity, nutrients, total suspended solids, conductivity, flushing flow frequencies, water and air temperatures, radiation, and sunshine hours for mainstem monitoring sites on the river were compiled with assistance from CCC.

   (b) Trends in the above data were analysed to identify any factor that might have stimulated increased weed growth.

   (c) Surficial sediment samples in the river were collected and analysed for nutrient content at survey cross-sections. This was done to evaluate the potential for sediment nutrient availability to influence nuisance weed growth. To the authors' knowledge there are no existing data of this kind for the river.

The Pūharakekenui/River Styx weed management investigation was carried out by NIWA from November 2020 to June 2022. The methodology, results and implications of the investigation are described in the sections below.

2 Methods

2.1 Delimitation of nuisance weed

2.1.1 Cross-section survey locations

Ten cross-sections along the length of the Pūharakekenui/Styx River between Redwood Springs and Kainga/Harbour Road were selected for carrying out a fine-scale assessment (Figure 2-1, Table 2-1).

Cross-sections were selected to provide a mixture of shaded and unshaded sites, good access from nearby roads, and were evenly distributed along the river length. Cross-section surveys were carried out on 9 November 2020, 2-3 February 2021, and 7 April 2021.

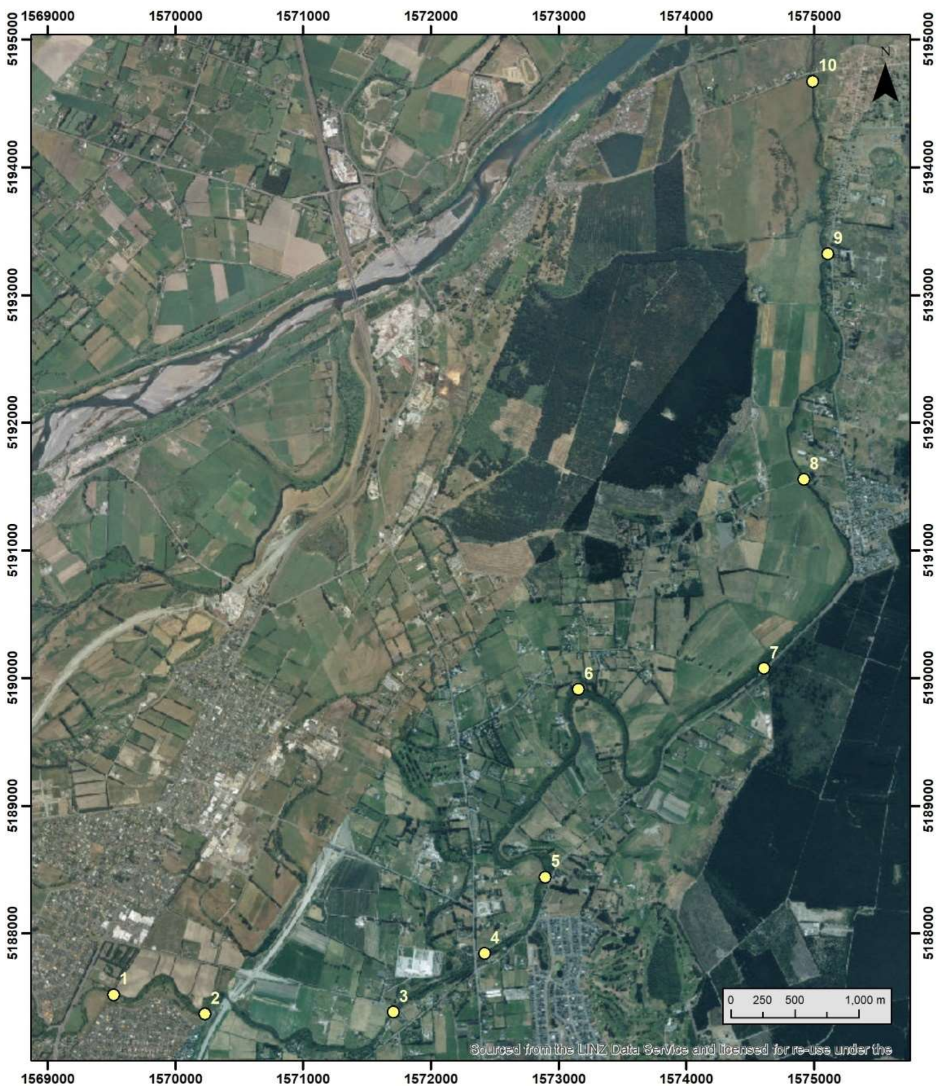

Figure 2-1: Cross-section survey location map. The map depicts the cross-section survey sites as referred to in the report text.

Table 2-1: Cross-section survey location details. Sites were selected to provide an even number of shaded versus open sites, alternating where possible, and at regular intervals.

Site number	Site name	Latitude	Longitude	Targeted shade level
1	Redwood Springs	43° 27.876'S	172° 37.389'E	Shaded
2	Willowview Drive	43° 27.960'S	172° 37.919'E	Shaded
3	Radcliffe Rd	43° 27.955'S	172° 39.014'E	Open
4	Janet Stewart Reserve[1]	43° 27.709'S	172° 39.545'E	Open
5	Dunlop Homestead	43° 27.386'S	172° 39.899'E	Open
6	Turners Loop	43° 26.592'S	172° 40.096'E	Shaded
7	Lower Styx Rd	43° 26.502'S	172° 41.175'E	Shaded
8	Spencerville Rd	43° 25.709'S	172° 41.410'E	Open
9	Boat Ramp Reserve	43° 24.753'S	172° 41.551'E	Shaded
10	Kainga/Harbour Rd	43° 24.024'S	172° 41.470'E	Open

[1] The cross-section at Janet Stewart Reserve was moved approximately 30 m downstream after the November 2021 survey due to construction activities next to the river in this area.

2.1.2 Cross-section survey measurements

At each cross-section, the channel wetted width was measured and five sampling points of 0.1 m^2 extent were marked using fence standards at 10, 30, 50, 70 and 90% of the width (Figure 2-2), in order to sample the full cross-section. At each sampling point the following measurements and semi-quantitative observations were made:

- Water depth (m).
- Maximum height of each aquatic plant species present (m).
- Percent cover of each aquatic plant species present (visually estimated to nearest 5%).
- Percent "clogginess" [4] of each aquatic plant species present (visually estimated to nearest 5%).
- Extent of canopy cover was estimated as an index of shade using a spherical densiometer (Forestry Suppliers Model C).

At the 50% (mid-channel) sampling point only, shade was also estimated using a clinometer following the protocol of Rutherford (2018). Canopy angles and bank angles to the channel mid-point were measured in eight directions, each 45°apart, canopy gap fraction was assessed and dominant vegetation type was noted as: 1) rank grass/weeds, 2) native shrubs/trees, 3) deciduous shrubs/trees or 4) other obstruction, such as bridge.

In addition, sediment and water samples were collected from each site to assess their risk of contributing to nuisance weed growth (expressed as low, medium (adequate) or high), according to Matheson et al. (2012).

[4] This is the percent occupation by the plant of a column of water with basal area equivalent to the quadrat size extending from river bed up to the water surface.

During each survey, and prior to the measurements described below, a water sample was collected in the centre of the channel at each site. All samples were chilled, and later analysed at the NIWA Hamilton water quality laboratory for nutrients. During the spring survey only, a 60 mL sampling syringe with nozzle removed was used to collect a small surficial sediment core (5 cm diameter x 2 cm depth) from sampling points at 10, 50 and 90% of the channel wetted width.

Sediment samples were analysed for total-N using an Elementar CN analyser (method: MAM, 01-1090), and total-P was determined after acid digestion by flow injection analysis of resulting ortho-phosphate (method: APHA 4500N/P (mod)). Water samples were analysed for nitrate-N, ammoniacal-N, and dissolved reactive-P using a Lachat flow injection analyser, conductivity by meter (APHA method 2510B), and turbidity with Hach TL2310 turbidimeter rated against formazin standards (APHA method 2130B).

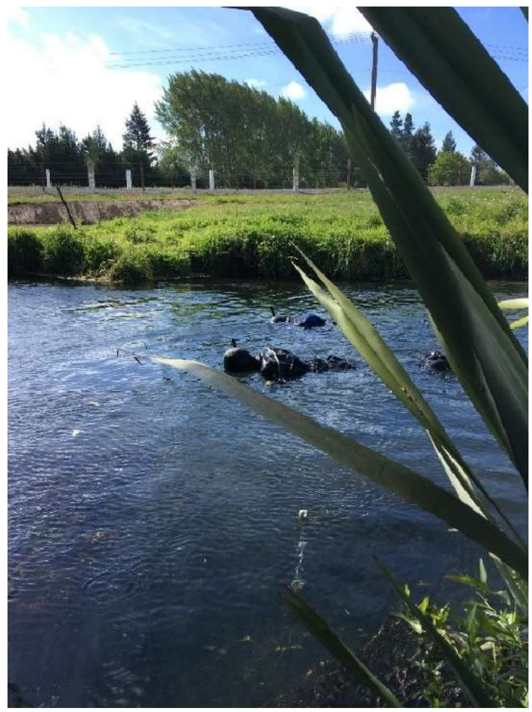

Figure 2-2: Cross-section surveys. Left: Willowview Drive cross-section. Right: snorkelers visually assessing weed clogginess at Janet Stewart Reserve cross-section.

2.1.3 Hydroacoustic surveys

Broad-scale surveys using hydroacoustics were performed seasonally to generate maps of aquatic plant biovolume (percent of water column occupied by submerged aquatic vegetation). These surveys were carried out immediately after the cross-section surveys. The maps provide an indication of weed nuisance.

The Pūharakekenui/Styx River was surveyed using this approach from just upstream of Radcliffe Road bridge downstream to Kainga/Harbour Road bridge. Encroaching marginal vegetation (mostly willows) prevented the transit of the small boat (required for the hydroacoustics survey) over a reach extending downstream from Redwood Springs to approximately 500 m upstream of Radcliffe Road.

Consequently, submerged aquatic vegetation in this reach was assessed at the cross-section survey points only (Sites 1 and 2, Figure 2-1, Table 2-1).

- Hydroacoustic surveys used a Lowrance™ depth sounder/GPS/chart plotter (models HDS 7 or 9, Navico Inc), Point-1 GPS Antenna (Lowrance™) and transducers (Total Scan) with dual frequency (200 and 455 kHz), deployed either on an unmanned surface vessel operated from the riverbank in an initial test-run in November 2020 (Figure 2-3), but subsequently, on a small, manned, motorised boat for February and April 2021 surveys.
- Sonar settings (ping speed, sensitivity and greyline) were locally optimised for bottom and vegetation detection. Digital data were logged as .Sl3 files.
- Files based on 200 kHz frequency were post-processed using BioBase (https://www.biobasemaps.com/) that utilises algorithms to extract GIS referenced bathymetry and vegetation biovolume for mapping. Vegetation biovolume estimated using BioBase is presented as a heat map and metrics on vegetation development as biovolume were extracted from BioBase vegetation reports.
- Geostatistical interpolated grid data are presented in this report, representing kriged (smoothed) output of aggregated data points (point data) using a 25 m buffer. Gridded data provide the most accurate summary of individual survey areas.

Hydroacoustic data were collected on each assessment date by traversing the river in an upstream direction, moving diagonally from bank to bank (Figure 2-3). Two limitations of the hydroacoustic mapping method are noted: Firstly, the hydroacoustic method requires water depths > 0.73 m, which is the minimum depth for accurate vegetation detection. Secondly, the acoustic signal can usually detect the bottom even in dense vegetation beds. However, where vegetation is extremely dense the acoustic signal detects the top of the vegetation as the stream bottom, thereby giving false depth and biovolume readings. During the hydroacoustic surveys, visual estimates of weed abundance and the dominant species present were made as the survey progressed upstream to enable validation of the hydroacoustic data.

Figure 2-3: Mapping submerged weed beds using hydroacoustics. Top: the hydroacoustic unit was deployed on NIWA's unmanned surface vessel in a trial run in November 2020. Bottom: Example of hydroacoustic data collection track. Map shows area upstream of Kainga/Harbour Road. Hydroacoustic data were collected on each assessment date by traversing the river from downstream to upstream moving diagonally from bank to bank. Direction of river flow from bottom to top of image.

2.2 Harvesting review

CityCare provided information on time periods in the 2020-21 season when the harvester was operated in the Pūharakekenui/Styx River, and estimates of weed removal volumes. The harvester operated in the river from 11 January to 21 February 2021.

For the 2020-21 harvesting season, the water level reduction achieved was assessed from water level records at Radcliffe Road and Lower Styx Road.

In addition, CCC provided harvesting start and end dates and estimates of water level reductions resulting from harvesting from June 2000 to May 2017. Using these data, the effectiveness of early season versus late season harvesting was assessed using a paired *t*-test with repeat measures for nominated harvested reaches. Water level reductions achieved in the upper versus mid- to lower river were compared using data provided for Radcliffe Road (upper river) and Lower Styx Road (mid- to lower river).

2.3 Diquat trial

Three potential trial sites were identified as suitable for the small-scale diquat application trial based on the following criteria:

- Occurrence of significant weed beds (determined via weed delimitation, section 2.1).
- Similar conditions present within a distance of approx. 300 m upstream, with no obvious inflows.
- Suitable access point for survey and sample collection.

The site selected was directly upstream of Radcliffe Road bridge because access was most straightforward, and its location furthest upstream ensured the least tidal influence on water movement (i.e., no upstream movement of applied diquat anticipated). This location corresponded to Site 3 in the delimitation surveys (section 2.1.1).

For the selected site at Radcliffe Road, an application reach, an upstream control reach and a downstream reach were defined (Figure 2-4).

Figure 2-4: Diquat trial - location of control, application, and downstream reaches at Radcliffe Road. Direction of river flow is from control reach to downstream reach.

On 21-22 March 2022, these reaches were surveyed to determine clogginess and cover of aquatic plants (total and by species) using the procedure described in section 2.1.2. The survey procedure was applied to five cross-sections, spaced at regular intervals, within each reach.

In the middle cross-section of each reach a datasonde (EXO multiparameter) was deployed in the centre of the channel to measure dissolved oxygen, water temperature, and water level at 15 min intervals for a three-month period. Additional probes were deployed on each instrument to measure the following: fluorescent dissolved organic matter (fDOM, a surrogate for coloured DOM), specific conductivity, turbidity, pH, and chlorophyll *a*. The datasonde was attached to a pole embedded in the river with instruments taking measurements in the water column 0.37 to 0.66 m above the river bed. Unfortunately, there was a battery issue with the sonde deployed at the application reach and the instrument stopped taking readings after 21 May 2022.

Figure 2-5: Surveying aquatic plants in the application reach. The sonde instrument (purple covering) is visible in the centre of the cross-section.

Diquat was applied to the target plants (elodea and curlyleaf pondweed) in the application reach by Boffa Miskell on 23 March 2022 from 1000 to 1230 (Figure 2-6). Diquat was applied directly onto the target weed beds in the reach via knapsack sprayer from a non-motorised vessel (Boffa Miskell 2022). The application was carried out under permitted activity Rules 5.22 and 5.163 of the Environment Canterbury Land and Water Regional Plan and a Certificate of Compliance CRC223323 was obtained for this work (Boffa Miskell 2022).

Diquat was applied at a rate calculated to achieve a maximum diquat concentration of 1 mg/L in the water occupying the reach. This rate was determined using cross-section data provided to Boffa Miskell by NIWA, from the delimitation surveys and water level determined at the time of application. Eleven litres of diquat dibromide concentrate (tradename Reglone) was applied to the reach (approx. 0.3 ha area). The reach was divided into four sections along its length, from 0-50 m, 50-100 m, 100-150 m and 150-200 m. The measurements of each section refer to its distance from the upstream limit of the application reach. Diquat was mixed at a rate of 2.2 L per 10 L of water. It was applied to each of the sections travelling in a downstream direction. In the first section, 0-50 m from the upstream limit, 4.4 L of (mixed) concentrate was applied. In all subsequent sections 2.2 L was applied (Boffa Miskell 2022).

Figure 2-6: Diquat being applied to the trial reach by Boffa Miskell. The 200 m trial reach was located directly upstream of the Radcliffe Road bridge. Application took place between 1000 and 1230 on 23 March 2022. Photograph from Boffa Miskell (2022).

NIWA collected water samples in 500 mL amber glass bottles from the downstream ends of the application and control reaches in time intervals of 2, 4, 6, 8, 12 and 24 hours after application was completed. The purpose of these samples was to check that the applied herbicide did not linger in the water and did not move upstream to affect the control reach (if there was a tidal backflow). Water samples were collected in the centre of the channel. Samples were kept in a dark and chilled environment and were analysed for diquat using Liquid Chromatography/Mass Spectrometry (LC-MS/MS) on 25 March by Eurofins ELS under contract to Hill Laboratories.

Aquatic plant surveys undertaken to determine clogginess and cover were repeated at each transect, four weeks after diquat application on 20-21 April 2022, and c. 3 months later on 13-14 June 2022. Datasondes were retrieved after the June surveys. The flow record for Radcliffe Road corresponding to the trial period was obtained from Environment Canterbury to enable consideration of effects associated with flushing flows.

Statistica software was used for statistical analysis of data. A repeated measures analysis of variance (ANOVA) was applied to the aquatic plant abundance data (total clogginess, and clogginess of key species) to identify significant effects of diquat application. Post-hoc Tukey HSD tests were applied for pairwise comparisons (where significant effects were indicated). Data did not require transformation prior to analysis.

2.4 Riparian shade evaluation

The Christchurch and Ashley River LiDAR 1 m DSM grid (2018-2019) was downloaded from the LINZ data service and combined with aerial photographs from 2018-19 in ArcGIS. The LiDAR data had a vertical accuracy specification of ± 0.2 m (95%). At 100 m intervals along the river, beginning at Redwood Springs and finishing at Kainga/Harbour Road bridge, transect lines were drawn across the channel wetted width. At each transect, at points 10, 30, 50, 70, and 90% of the wetted width, shade levels were estimated using ArcGIS software tools.

The ArcGIS Solar Radiation graphics tool was used to create viewshed graphics for each point. The viewshed graphic generated from the tool was analysed to calculate percent shade values for each point. Shade cannot be estimated by the ArcGIS tool when overhanging riparian vegetation is present, so these points were assigned a value of 90% shade based on estimates for unbroken canopy cover (Figure 2-7). For each transect, the five point values were averaged to provide a single estimate of shade. The same procedure was applied to estimate shade at additional transects applied at the field survey cross-section sites (section 2.1.2).

At each transect, riparian vegetation height on each bank was estimated. To do this, transect lines were extended to each side of the channel by a length equal to the width of channel at that point – the assumption being that vegetation beyond a channel width from the bank would be unlikely to control the canopy angle. Sampling points were created at 1 m distances along the length of the transect extension lines. Elevation values were extracted for these points from the LiDAR 1 m DSM grid. Elevation values from 1 to 5 m of the channel were averaged to provide an estimate of riparian vegetation height on each bank.

For each transect the ratio of estimated riparian vegetation height to channel wetted width was calculated, after an average of the riparian vegetation height was determined for each bank. Where a ratio of 1 or more occurred, shading of ≥70% was assumed, based on simple modelling presented by Davies-Colley and Rutherford (2005). This level of shading was considered sufficient to limit nuisance aquatic weed growth, according to Matheson et al. (2018). Where the ratio was less than 1, the additional vegetation height required to achieve a ratio of 1 was calculated.

Figure 2-7: Riparian shade modelling transects spaced at 100 m intervals along the length of the river. Left: Modelling points between Radcliffe Road and Dunlop Road. Right: Modelling transects and points downstream of Willowview Drive. The ArcGIS radiation calculator was used for relatively open points (white), but where vegetation was overhanging (green-highlighted points) the radiation tool does not work, so shade was set to 90%.

2.5 Water quality, sediment, and climate effects

2.5.1 Data collation

Hydrometric, water quality and climate data for the period 1 January 2007 to 31 December 2019 were compiled with assistance from CCC and NIWA staff.

Daily water level and flow data were obtained for Radcliffe Road (level and flow), Lower Styx Road (level only) and Kainga/Harbour Road (level only) from CCC. From these records, daily baseflow at Radcliffe Road was calculated using the Lyne and Hollick digital filter method (Ladson et al. 2013). The number of days since the last flow event equivalent to 2, 3 and 4 times the median flow for the record was also calculated.

Monthly water quality records of water temperature, conductivity, suspended solids, nitrate-N, dissolved inorganic-N, total-N, dissolved reactive-P and total-P for Marshlands Road, Richards Bridge and Kainga/Harbour Road were provided by CCC. Salinity was calculated from records of conductivity and water temperature according to UNESCO (1983) and Hill et al. (1986).

Daily sunshine hours, radiation, and air temperatures for the nearest climate monitoring station (Christchurch Airport) were downloaded from NIWA's Aquarius database. Only daily minimum and maximum air temperature data were available for the entire record.

2.5.2 Trends analysis

A trends analysis was conducted for all key parameters to ascertain if an increasing or decreasing trend was evident through time, and the confidence in that trend was determined. Data from 1 July 2007 to 30 June 2019 was analysed in order to begin the analysis in the annual period of winter dormancy for aquatic plants, and to avoid using baseflow values close to the beginning of the calculated record, which are less reliable.

Trends were assessed following the trend direction assessment (TDA) procedure outlined in McBride (2019) and implemented using the LWP-Trends library in R (Snelder and Fraser 2019, Larned et al. 2018). In this method, trend direction is determined using Kendall S statistics and p-values. The Kendall test is a nonparametric correlation coefficient which measures the monotonic (single direction) association between a variable y and, in temporal trend analysis, time x. The Kendall p-value can be interpreted as the probability that the trend is decreasing by:

$$P(S<0) = 1 - 0.5 \times pvalue \qquad\qquad P(S>0) = 0.5 \times pvalue$$

Where pvalue is the p-value returned by a seasonal or non-seasonal Kendall test, S is the S statistic returned by the Kendall test, and P is the probability that the trend was decreasing (Fraser and Snelder 2018). The trend direction is then interpreted as decreasing when $P > 0.5$ and increasing when $P < 0.5$ (Fraser and Snelder 2018).

If there were significant differences in measurements between seasons, a seasonal Kendall S statistic was calculated. This requires first calculating the S statistic for data from each season individually and then summing over all seasons (Fraser et al. 2021). The probabilities derived from the Kendall test statistics were then used to assign a confidence in trend direction category to each trend following the framework in Table 2-2.

Positive and negative trend direction results are complementary, i.e., Prob(positive trend) = 1-Prob(negative trend). Therefore, a "very likely" positive (or increasing) trend is the same as a "very unlikely" negative (or decreasing) trend (McBride 2019, also see Table 2-2). A LOESS (Local polynomial regression) or GAM (Generalised Additive Model) smoothing function was also applied to look for non-monotonic patterns in the data for comparison. The GAM smoother was applied where data points exceeded 1000.

Table 2-2: Trend analysis confidence categories assigned to Kendall test probabilities. Adapted from McBride (2019) and Mastrandrea et al. (2010), with original Intergovernmental Panel on Climate Change (IPCC) categories listed first in italics and the modified descriptions used in this study listed below them in bold.

Categorical level of confidence	Probability of increasing trend (%)
Virtually certain **Virtually certain decreasing**	99-100
Extremely likely **Extremely likely decreasing**	95-99
Very likely **Very likely decreasing**	90-95
Likely **Likely decreasing**	67-90
As likely as not **As likely increasing as decreasing**	33-67
Unlikely **Likely increasing**	10-33
Very unlikely **Very likely increasing**	5-10
Extremely unlikely **Extremely likely increasing**	1-5
Exceptionally unlikely **Virtually certain increasing**	0-1

3 Results

3.1 Delimitation of weed nuisance

3.1.1 Broad-scale mapping

The hydroacoustic surveys of the river in spring (November 2020), summer (February 2021) and autumn (April 2021) covered c. 31 hectares in area and the estimated waterbody volume ranged from 256,352 to 276,181 m^3. Water level between surveys varied for the upstream reach (Radcliffe Road) by 0.18 m and the downstream reach (Lower Styx Road) by 0.27 m. The highest water level was recorded for the autumn survey. No adjustment (offset) was deemed necessary in processing the hydroacoustic data.

An example of the output maps produced for one key segment of the river is shown below (Figure 3-1). Maps for all other river segments are provided in Appendix A. Areas shaded blue have low submerged vegetation biovolume, green have moderate biovolume, and yellow-red have high biovolume.

Overall, weed biovolume was larger in autumn than in spring and summer (Table 3-1). Biovolumes measured in summer (February) have been moderated by weed harvesting which occurred at this time. In autumn, the most extensive areas of high biovolume weed beds occurred between Site 6 (Turners Loop) and Site 8 (Spencerville Rd), particularly in the area known as the "Lower Styx Straight" which is included in Figure 3-1.

Table 3-1: Summary statistics from the seasonal hydroacoustics surveys. Vegetation biovolume values are means ± standard deviation.

Season/month	Avg BVw (%)[1]	Avg BVp (%)[2]	Percent cover (%)[3]
Spring/November	14.6 ± 11.7	17.4% ± 10.7	84.0
Summer/February (harvest-affected)	14.2 ± 13.1	17.6 ± 12.3	80.4
Autumn/April	30.2 ± 16.6	30.7 ± 16.2	98.4

[1] Avg BVw (%) = Biovolume (All water): Average percentage of the water column taken up by vegetation regardless of whether vegetation exists.
[2] Avg BVp (%) = Biovolume (Plant): Average percentage of the water column taken up by vegetation when vegetation exists.
[3] Percent cover (%) = Percent area covered by submerged vegetation over the survey area.

3.1.2 Fine-scale assessment

Cross-section surveys of aquatic plants within the wetted width of the river were carried out by snorkeling at ten sites, in spring, summer and autumn. The maximum water depth recorded during the surveys was 3.2 m at cross-section 10 in the lower river, just upstream of Kainga/Harbour Road and the tide gates.

Ten taxa/species of aquatic plants, a mixture of exotic and native species, were identified. These are listed in order of prevalence as follows: elodea, curlyleaf pondweed, stonewort (*Nitella* sp. aff. *cristata,* native), blunt pondweed (*Potamogeton ochreatus,* native), water milfoil (*Myriophyllum propinquum*, native), waterfern (*Azolla* spp.), duckweed (*Lemna minor*), water speedwell (*Veronica anagallis-aquatica*), watercress (*Nasturtium* spp.), starwort (*Callitriche stagnalis*) and crystalwort (*Riccia fluitans*, native). Some example photographs are provided in Figure 3-2 .

Figure 3-1: Weed biovolume maps for the fifth segment of the Pūharakekenui/Styx River downstream of Redwood Springs. River segments from Lower Styx Road to Spencerville Road Bridge in spring (November) 2020 (left), summer (February) 2021 (centre) and autumn (April) 2021 (right). "Lower Styx Straight" area is shown in lower third of each map. Biovolume (% of water column occupied by weed) key shown on the right.

Figure 3-2: Aquatic plant species in the Pūharakekenui/Styx River. Left: Elodea (light green) and stonewort (dark green) in a metal survey quadrat. Centre: Bed of surface-reaching curlyleaf pondweed with chironomid grazing damage. Right: the native milfoil, *Myriophyllum propinquum* in the foreground, with a bed of elodea behind in deeper water.

Total aquatic plant “clogginess”, or the percentage of water column occupied by aquatic plants in the surveyed cross-sections, ranged from 7% to 60% (mean 25%) across the three surveys (Figure 3-3). Average (± standard error, SE) clogginess increased from 19 (±2) to 26 (±4) to 29 (±6)% from spring to summer to autumn. At cross-sections 7 and 10, lower clogginess in summer compared to spring may be due to harvesting.

The native blunt pondweed (*Potamogeton ochreatus*) was present at low to moderate clogginess (up to 9-10%) in the lower part of the river (e.g., Sites 9 and 10). The native charophyte, *Nitella* aff. *cristata*, was also present at most cross-sections at low to moderate abundance (up to 10% clogginess).

Curlyleaf pondweed tended to be more abundant in shallow waters near the river margins, while elodea, stonewort and blunt pondweed often occupied the deeper sections of the river (Figure 3-4). Maximum and average water depths for the surveyed cross-sections were 3.2 m and 1.0 m, respectively. Average (± SE) water depths (m) for each of the four main species were: 0.92 (±0.06), 1.04 (±0.05), 1.03 (±0.06) and 1.32 (±0.24) for curlyleaf pondweed, elodea, stonewort, and blunt pondweed, respectively.

Freshwater mussels were observed at several sites (Figure 3-5) and were especially abundant at Site 7 (Lower Styx Road). In autumn (April 2021), curlyleaf pondweed specimens at some sites, especially upstream of Radcliffe and Dunlop Road Sites, showed significant leaf defoliation caused by chironomids (Figure 3-2), however stem material remained healthy. This leaf damage was not recorded in the earlier season surveys. In spring, weed beds were often heavily coated in algae, but not in summer or autumn.

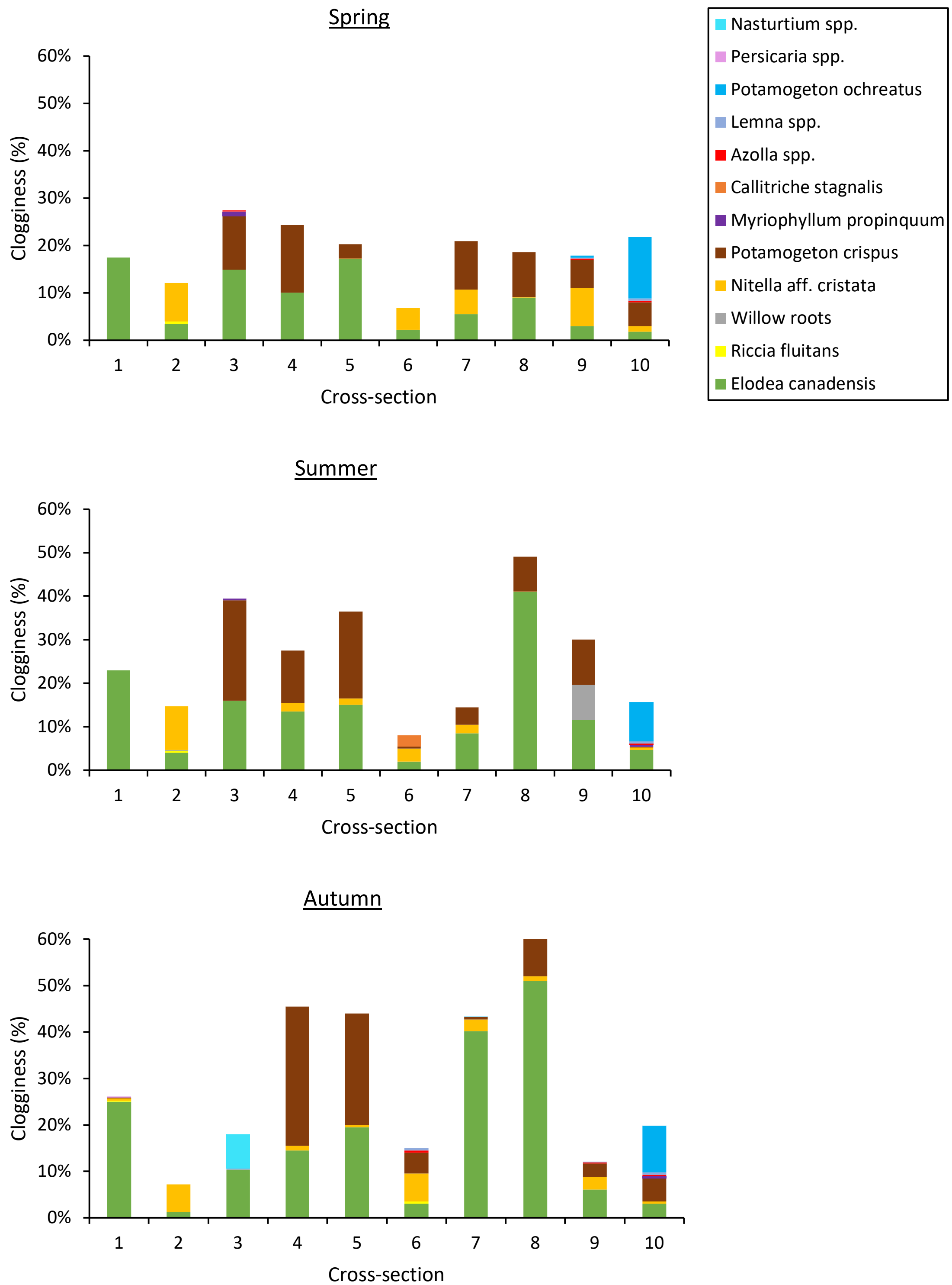

Figure 3-3: Channel clogginess of aquatic plant species at cross-sections by season. Clogginess for each species/taxa shown. Values for each species are means of five cross-section sampling points. Clogginess is the percentage of water column occupied by aquatic plants. Overall, bar length indicates the clogginess of all species combined.

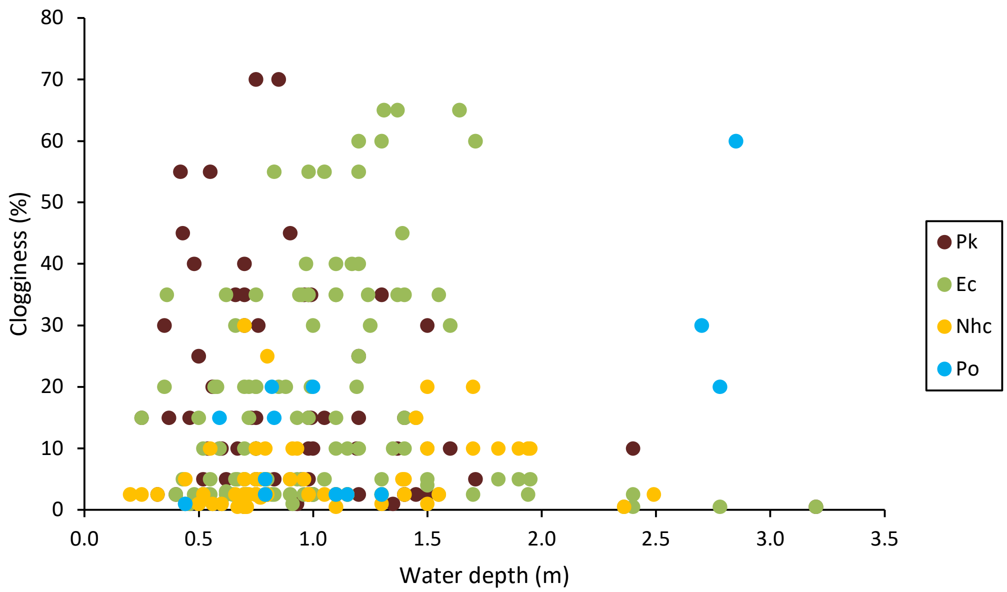

Figure 3-4: Clogginess records for the four main aquatic plant species by water depth. Pk = *Potamogeton crispus* or curlyleaf pondweed, Ec = *Elodea canadensis* or elodea, Nhc = *Nitella* sp. aff. *cristata* or stonewort, Po = *Potamogeton ochreatus*, or blunt pondweed. A total of 150 Records shown for all five points (i.e., 10, 30, 50, 70 and 90% of wetted width) at each of 10 surveyed river cross-sections in spring, summer, and autumn.

Figure 3-5: Freshwater kākahi and weed beds covered with algae in spring. Kākahi (or freshwater mussels, kāeo, torewai) (left), and algae-coated weed beds (right), both at cross-section survey Site 3, Radcliffe Road.

3.2 Harvesting review

3.2.1 2020-21 season

CityCare estimates of weed removal show that around 140 tonnes of wet weed was extracted from the river by the harvesting operation, and deposited in piles adjacent to the river (Figure 3-7), in the period from 11 January to 21 February 2021 (Table 3-2).

Table 3-2: Estimates of weed removed by harvesting in summer 2021. Data supplied by Citycare.

Starting date	Area of operation	Tonnes of weed removed
11 January	Spencerville corner and straight	39
19 January	Radcliffe Rd bridge downstream to Muschamps farm[1]	41
21 January	Muschamps farm[1] to Earlham St bridge	30
12 February	Earlham to floodgates	30
Sum	**All**	**140**

[1] Lower Styx Road

Hydroacoustic and cross-section data indicated significant accrual of weed between February and April (Table 3-1, Figure 3-3), despite the harvesting operation removing a substantial quantity of weed in summer. This suggests the effect of harvesting when conducted in February is short-lived owing to re-growth.

Water level records (to R.L.) for Radcliffe Road indicated that the 2021 harvesting operation may have reduced water levels in the river by c. 0.16 m. When harvesting was initiated on 11 January, the water level was 11.1 m and peaked at 11.21 m on 17 January (Figure 3-6). It then gradually dropped to a low of 10.94 m, four days after the harvesting operation was completed, on 25 February.

At Lower Styx Rd the potential effect of harvesting on water level was more rapid and of similar magnitude, but not sustained. When harvesting was initiated on 11 January, the water level was 10.10 m and it dropped to a low of 9.83 m on 29 January (Figure 3-6). It then gradually increased back to 9.92 m when harvesting ceased on 21 February. A maximum potential reduction of 0.27 m was indicated by the 11-19 January data, but a reduction of only 0.18 m at harvesting completion on 21 February, most likely reflecting rapid regrowth.

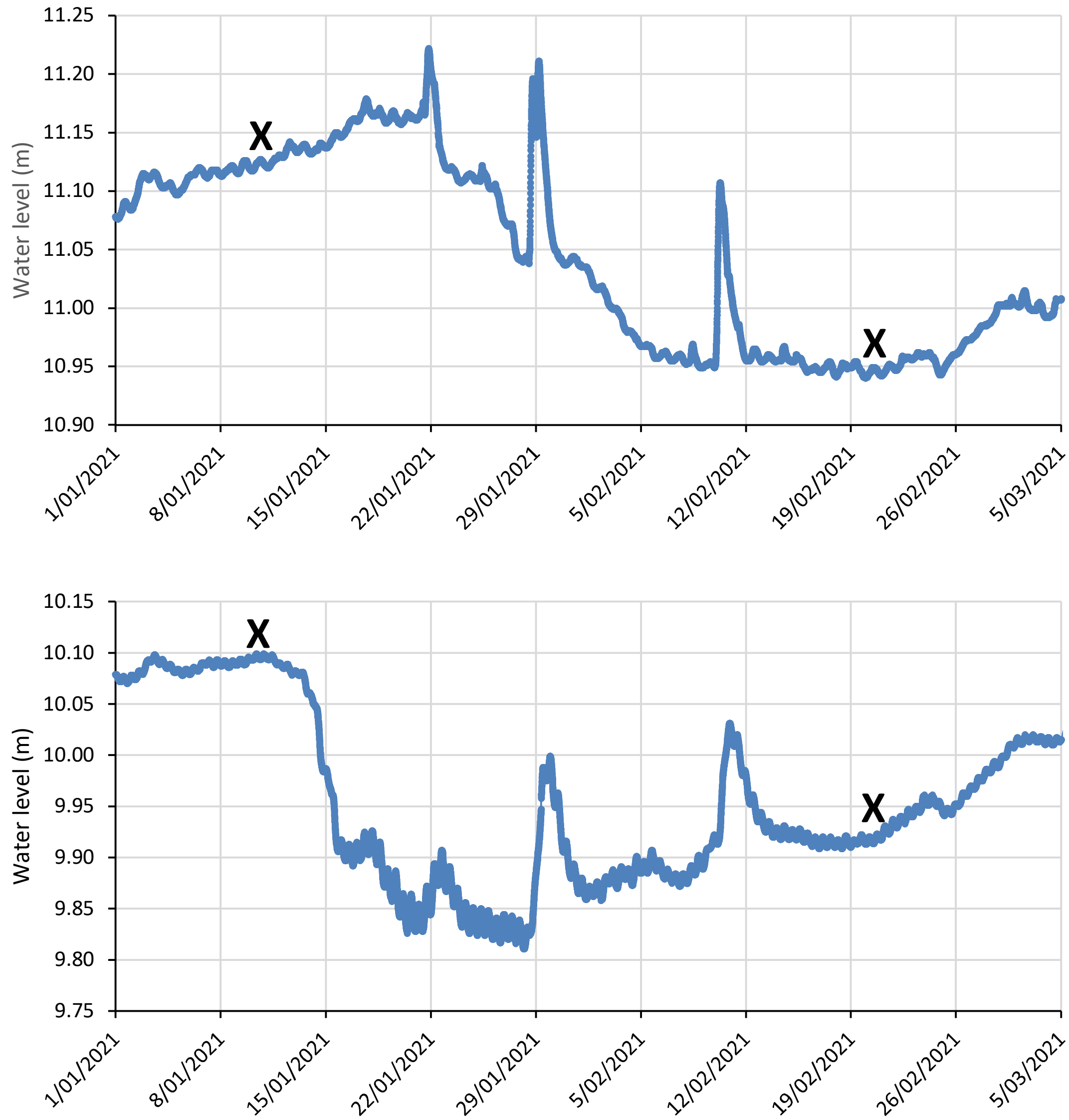

Figure 3-6: Pūharakekenui/Styx River water level during summer 2021. Top: Radcliffe Road. Bottom: Lower Styx Road. Black crosses indicate weed harvesting start and end dates. Three spates, on 23 January, 30 January, and 10 February, which temporarily raised water levels, are evident in the records for both sites. Water level records from 1 July 2020 to 30 April 2021 are provided in Appendix B.

During the cross-section surveys and hydroacoustic mapping in early February 2021, the field team noted that harvested weed was deposited in regular piles along the river margins, from downstream of Marshland Road to upstream of Spencerville Road (Figure 3-7).

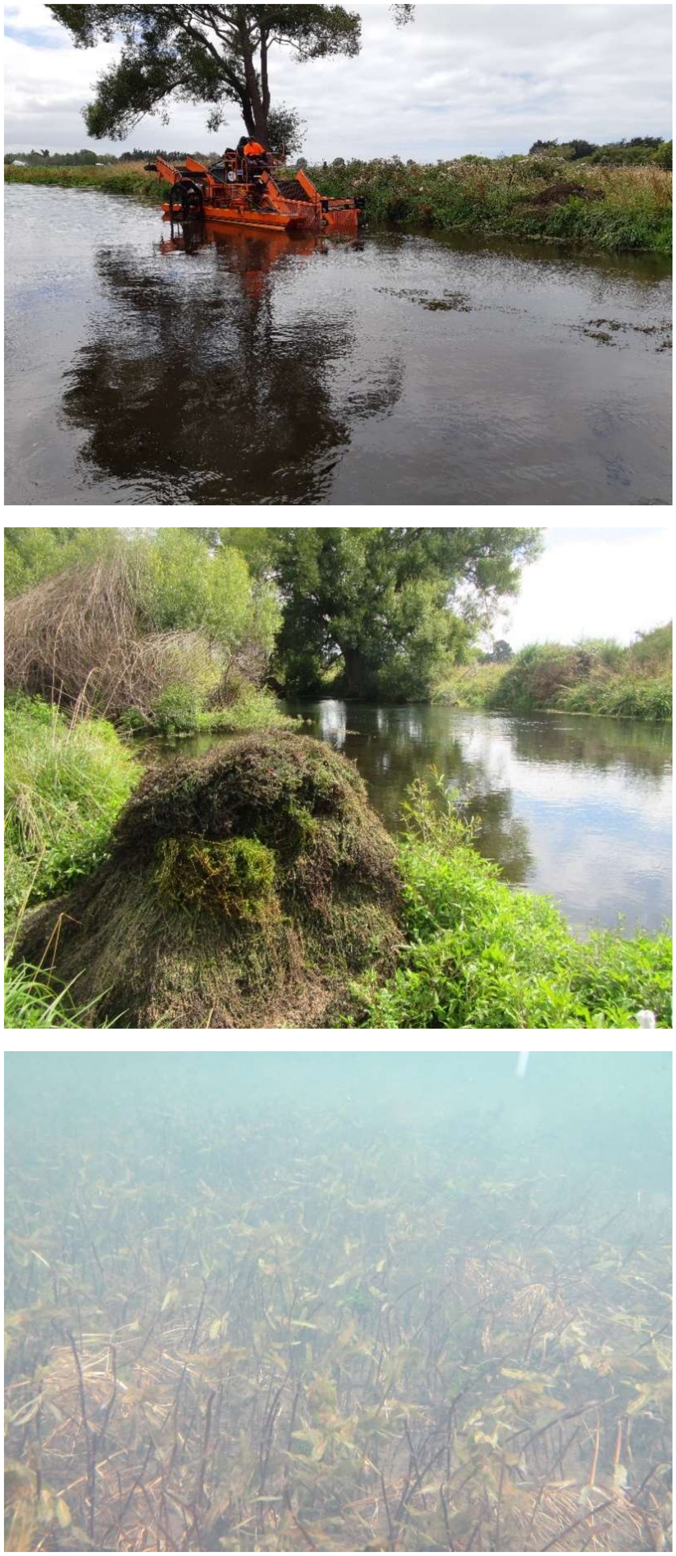

Figure 3-7: Weed harvesting in the Pūharakekenui/River Styx - February 2021. Top: Weed harvester operating during the NIWA February survey. Middle: Harvested piles of weed deposited in mounds on the riverbanks (mostly the exotic elodea in the photo shown). Bottom: A harvested stand of exotic curlyleaf pondweed showing clipped brown stems of taller plants and short-stature healthy regrowth.

3.2.2 2000-2017 water level reductions

Water level changes associated with harvesting have been recorded by CCC from 2000 to 2017. In the last ten years (2007 to 2017) these reductions have ranged from -0.25 m (i.e., a 0.25 m water level increase) to +0.73 m (i.e., a 0.73 m water level decrease). When plotted, the reductions show a high degree of variability (Figure 3-8). Statistical analysis indicates a likely increasing trend in water level reductions achieved with harvesting at Lower Styx Road and a likely decreasing trend for Radcliffe Road. This suggests that harvesting has become somewhat more effective in the lower part of the river over time (i.e., achieved greater reduction in water level), and less effective in the upper part of the river (i.e., increased water level).

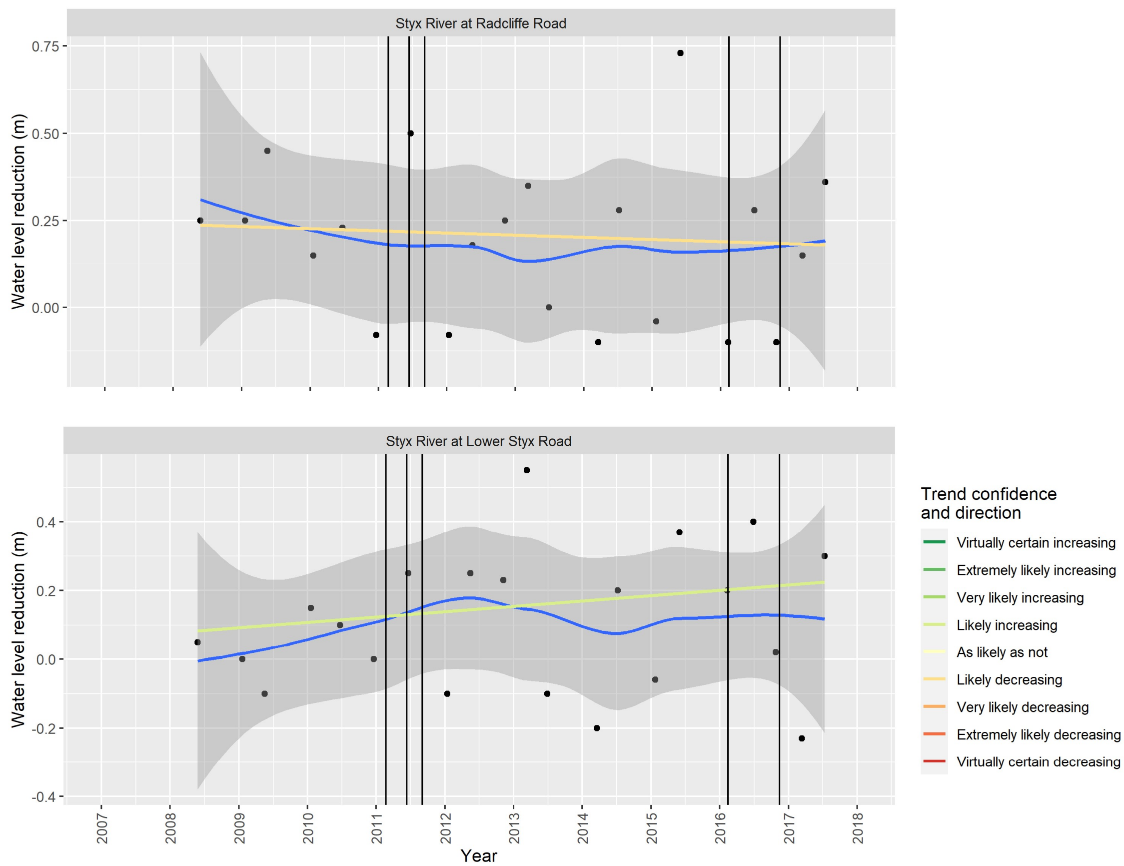

Figure 3-8: Trends in water level reductions associated with harvesting from 2007 to 2017. Top graph: Radcliffe Rd. Bottom graph: Lower Styx Rd. Raw data supplied by Christchurch City Council. Black vertical lines indicate significant earthquake occurrences. Blue line is a local polynomial regression (LOESS) curve fit with shaded band indicating the 95% confidence interval.

Harvesting can be triggered, and carried out, early or later in the growing season. September to December is the spring to early summer initial growth period for aquatic plants, while the period from January through to April is the second half of the growing season, and often the period when highest weed biomass accumulates.

According to CCC records, in all years there were two harvesting periods: an early harvest, often in November, and one later. However, in one year (2006) the first harvest occurred in February and in other years (2013, 2017) a third harvest was performed late in the season (see Appendix B for further details).

Statistical analysis indicates that water level reductions achieved with harvesting carried out later in the season (e.g., January to May, in general) were significantly greater (more successful) than those obtained from operations earlier in the season (September to December, in general) (paired *t*-test, $p<0.05$) (Figure 3-9, Table B-1).

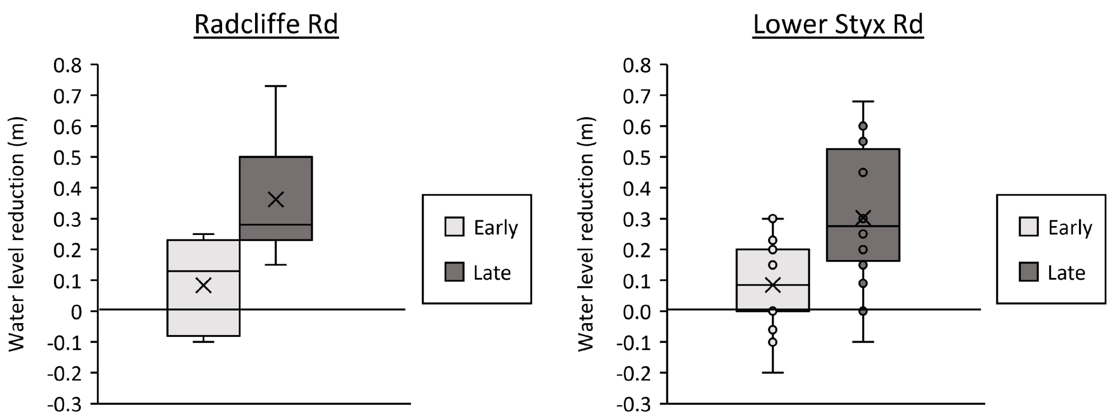

Figure 3-9: Water level reductions associated with early and late season harvesting at Radcliffe Rd and Lower Styx Rd. Raw data from Christchurch City Council (2000 to 2017). Boxplots show mean (-), median (x), interquartile range (box) and variability outside interquartile range (whiskers, |-|). Late season water level reductions are significantly higher than those earlier in the season (paired *t*-test, $p<0.05$).

3.3 Diquat trial

3.3.1 Diquat concentrations

Diquat was below detection limits (0.001 mg/L) in water samples collected from the downstream end of the application reach, and in water samples collected from the downstream end of the control reach in the period from 2 to 24 hours of application. This suggests that the diquat dispersed relatively quickly and/or was rapidly adsorbed to target plants or clay and organic constituents in water or sediment.

3.3.2 Effect of diquat on aquatic plants

The surveys showed that the effect of diquat on total clogginess and cover of aquatic plants, and the clogginess and cover of key species, in the application reach, and at a second reach 1 km downstream, was small to negligible (Figure 3-10), and was not statistically significant (RM ANOVA, reach/reach x time, $p>0.05$[5]). Plots showing the cover data are provided in Appendix C (Figure C-1).

[5] A significant effect among reaches was observed for total clogginess $p = 0.025$ and Ec clogginess $p=0.005$ when the RM ANOVA was applied, but upon post-hoc testing *p* values among pair-wise comparisons then exceeded 0.05 and were not significant.

In all three survey reaches clogginess decreased significantly over time (RM ANOVA, time, $p < 0.05$), consistent with seasonal senescence of the aquatic plant biomass and dislodgement associated with occasional spate/flood flows. A photo time-series from the surveys is provided in Appendix C (Figure C-2).

Using the control reach as the benchmark indicative of conditions unrelated to diquat application, total clogginess in this reach reduced by an absolute percentage of 28% from 21 March to 13 June. In comparison, clogginess in the application reach reduced over the same period by a slightly higher margin of 35% and in the downstream reach by an equivalent amount as the control reach (28%).

A similar result was evident when data for only elodea was considered, this being the main weed species present in the control and application reaches, and the primary target. For this species, in surveys one month after diquat application, divers noted loss of some apical shoots in the application reach and they also reported blackened stems of elodea in several application plots (see Appendix C, Figure C-2. for photo time series from cross-section plots). The latter is consistent with defoliation caused by diquat. However, although there was a reduction in elodea clogginess at the application and downstream reaches after diquat application, clogginess also declined at the upstream control reach, and upon statistical analysis of the data the effect of diquat was found to be insignificant.

Curlyleaf pondweed, the other weed species in the river, was more abundant in the downstream reach than in the other two reaches upon initial survey. Like elodea, clogginess of this species also decreased after diquat application but at all three reaches, and the diquat effect was not significant.

Clogginess of the main native species, common water milfoil and stonewort was less than 10% at all reaches throughout the trial period. Some slight increases in clogginess of these species after diquat application were indicated by the data, but these changes did not register as statistically significant.

Clogginess of all other species, which were minor components of the aquatic plant flora, also showed no pattern of decline that could be attributed to the diquat application.

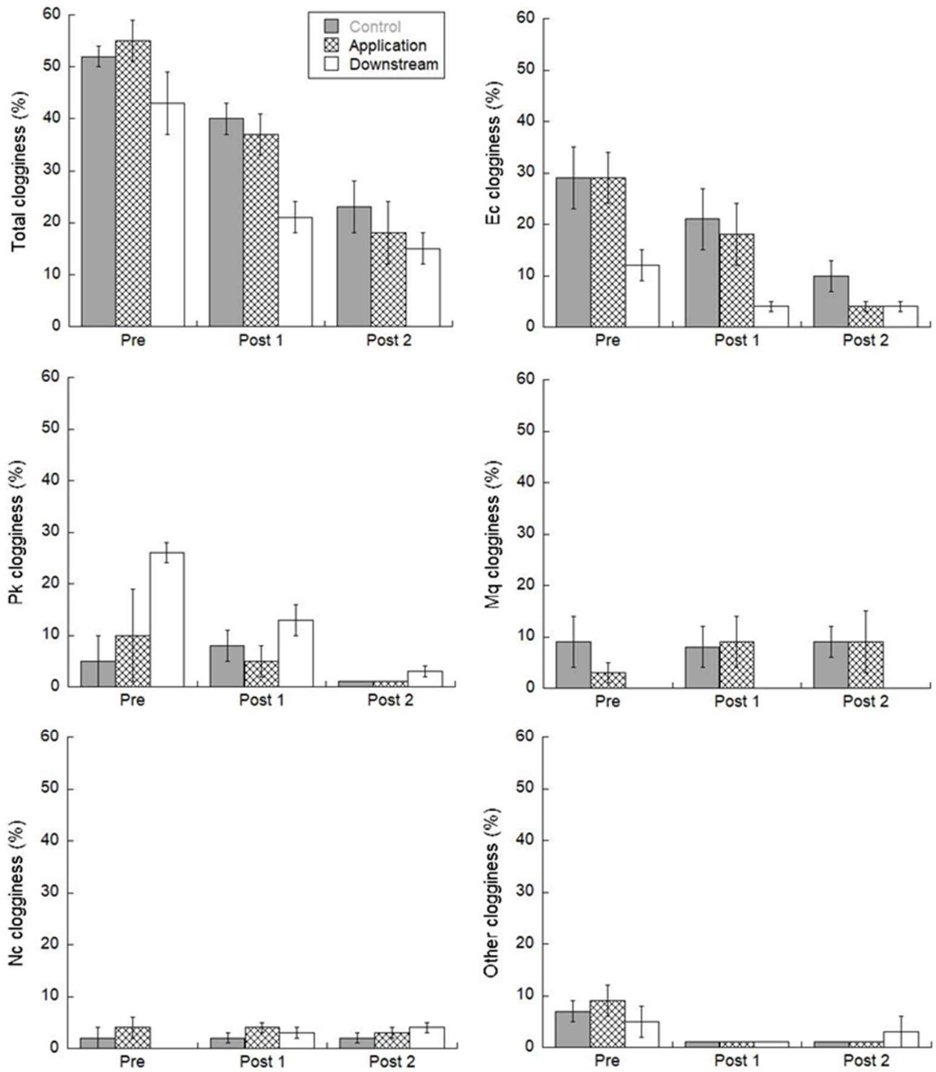

Figure 3-10: Clogginess of aquatic plants, before and after diquat application in control, application, and downstream reaches. Values shown are means with bars representing the standard error (n=5). Pre = 21-22 March 2022 survey, Post 1 = 20-21 April 2022 survey (4 weeks after treatment), Post 2 = 13-14 June 2022 survey (12 weeks after treatment). On y-axes, Ec = *Elodea canadensis*, elodea. Pk = *Potamogeton crispus*, curlyleaf pondweed. Mq = *Myriophyllum propinquum*, common water milfoil. Nc = *Nitella* aff. *cristata*, stonewort. Other = all other aquatic plant species present.

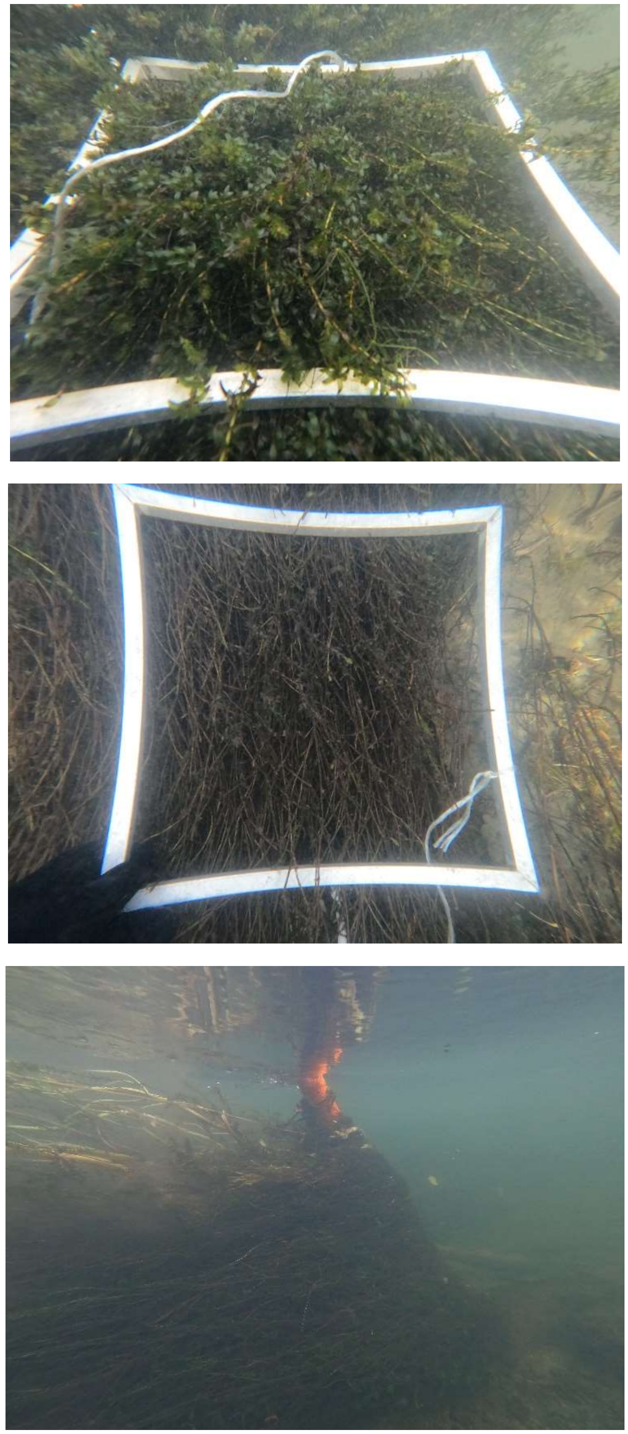

Figure 3-11: Selected images from the aquatic plant surveys one month after diquat application. Top: bed of elodea in the application reach where some loss of apical shoots likely caused by diquat was noted by divers. Centre: Blackened stems of elodea detected in several plots in the application reach are indicative of defoliation caused by diquat. Bottom: significant weed drift snagged around the sonde instrument (wrapped in orange tape) in the upstream control reach.

3.3.3 River stage, flows and water quality

River stage and flow was elevated on the day scheduled for diquat application (23 March 2022) due to rain that began the preceding day, but with application taking place as peak flow receded (Figure 3-12). Although stable flow conditions are preferable, weed and water conditions were considered suitable in the weeks leading up to the application, and on the day of application.

Table 3-3: River stage and flow during aquatic plant surveys and diquat application. Data from Environment Canterbury monitoring station at Radcliffe Road. The station is located at the road bridge, just downstream of the diquat application reach. River stage data from Environment Canterbury, accessed July 2022, records were provisional from 5 May 2022. Arrows indicate increase or decrease in stage or flow during each event (time period defined in column 2)

Event	Dates	River stage (m)	River flow (m^3/s)
Pre survey	21-22 March 2022	11.404 to 11.653 (↑)	1.359 to 2.379 (↑)
Diquat application	23 March 2022 (1000-1230)	11.479 to 11.472 (↓)	1.632 to 1.605 (↓)
Post survey 1	20-21 April 2022	10.971 to 10.981 (↑)	1.494 to 1.490 (↓)
Post survey 2	13 to 14 June 2022	10.698 to 10.669 (↓)	1.563 to 1.458 (↓)

Diquat can be adsorbed to clay particles and organic matter suspended in water. However, at the time of diquat application, the sonde measurements suggested that concentrations of waterborne fDOM (coloured dissolved organic matter), turbidity and chlorophyll *a* in the application reach were relatively low at 48 QSU, 1.4 FNU and 1.8 µg/L, respectively.

In addition to the flow peak on day of application, several other flow spikes were evident in the record for the trial period. The largest three of these occurred on 7 April, 17 May, and 4 June (Figure 3-12). These elevated flows were likely to have mobilised sediments and dislodged plant material, especially senescent material affected by seasonal die-back or due to diquat application. Weed drift was noted in the river with material snagged around posts deployed for the sonde instruments in all three reaches during the trial period.

River stage at Radcliffe Road dropped during the trial period from a peak of 11.653 m on 21-22 March to 10.669 m on 13-14 June (see Appendix C, Figure C-3 for stage plot), consistent with anecdotal reports of low levels from CCC staff. However, river flow records indicate relatively stable discharge over the same period (Figure 3-12) and are consistent with the water depth data recorded by sonde in the diquat application reach, also located at Radcliffe Road (Figure 3-13). Water depth data from the sondes at the control and downstream reaches show declining trends during the trial period. However, at reaches we note that the sonde water depth data might have been affected by drift accumulation altering pole position (see Figure 3-11). Water depths measured by divers during the aquatic plant surveys show lower water levels at all reaches in the two assessments that occurred after diquat application (Table 3-4). Declining river stage associated with little change in discharge is consistent with removal of weed in the river and the impediment to flow and increase in water stage/level that it can cause (Champion and Tanner 2000). This aligns with our observations of declining weed clogginess during the trial period.

Water depth measured by divers at each reach varied through time (Table 3-4), but generally in a manner consistent with the stage record for Radcliffe Road (see Figure C-3). The lowest recorded water depths were in the latter part of the trial, notably at the downstream reach. The 13-14 June surveys recorded maximum water depth at the application reach of 120 cm, down from 179 cm at the beginning of the trial.

No adjustments were applied to clogginess data (a measure of the water depth occupied by aquatic plants) in response to water depth differences among survey dates. This was not considered necessary as water depth (recorded by divers) in all three reaches decreased over time in a consistent manner.

Table 3-4: Water depths measured during the aquatic plant surveys. Values in parentheses are means.

Event	Measured water depth range (cm)		
	Control reach	Application reach	Downstream reach
Pre survey	28-206 (92)	18-179 (101)	78-206 (246)
Post survey 1	1-125 (46)	1-138 (61)	1-164 (80)
Post survey 2	1-104 (38)	1-120 (46)	1-125 (68)

Higher concentrations of fDOM, and lower levels of specific conductivity, were associated with elevated flows (Figure 3-12). These concentration changes indicate periodic flushes of storm runoff into the river.

The causes of elevated levels of specific conductivity and turbidity at the downstream reach in the latter part of the trial (from c. 14 May 2022 in particular) are unknown.

It is possible that a set of elevated chlorophyll *a* concentrations in the application reach approximately 7 days after diquat application are indicative of weed breakdown. However, periodic spikes of chlorophyll *a* were apparent in the record for all three reaches and the origin of these is unknown (Figure 3-13).

Dissolved oxygen followed a consistent diurnal pattern at all three reaches until the 3 May 2022, after which concentrations at the downstream reach were irregular and often low (Figure 3-14). As noted above, turbidity, specific conductivity, fDOM and pH similarly all showed indications of disturbance at the downstream reach during this period of the record. There is no indication of dissolved oxygen depletion occurring as a result of the diquat application.

Water temperature records were very consistent among the three reaches and tracked downwards with the seasonal shift from autumn to winter conditions. Water temperatures reduced from c. 16°C at the beginning of the trial to 11°C at trial completion.

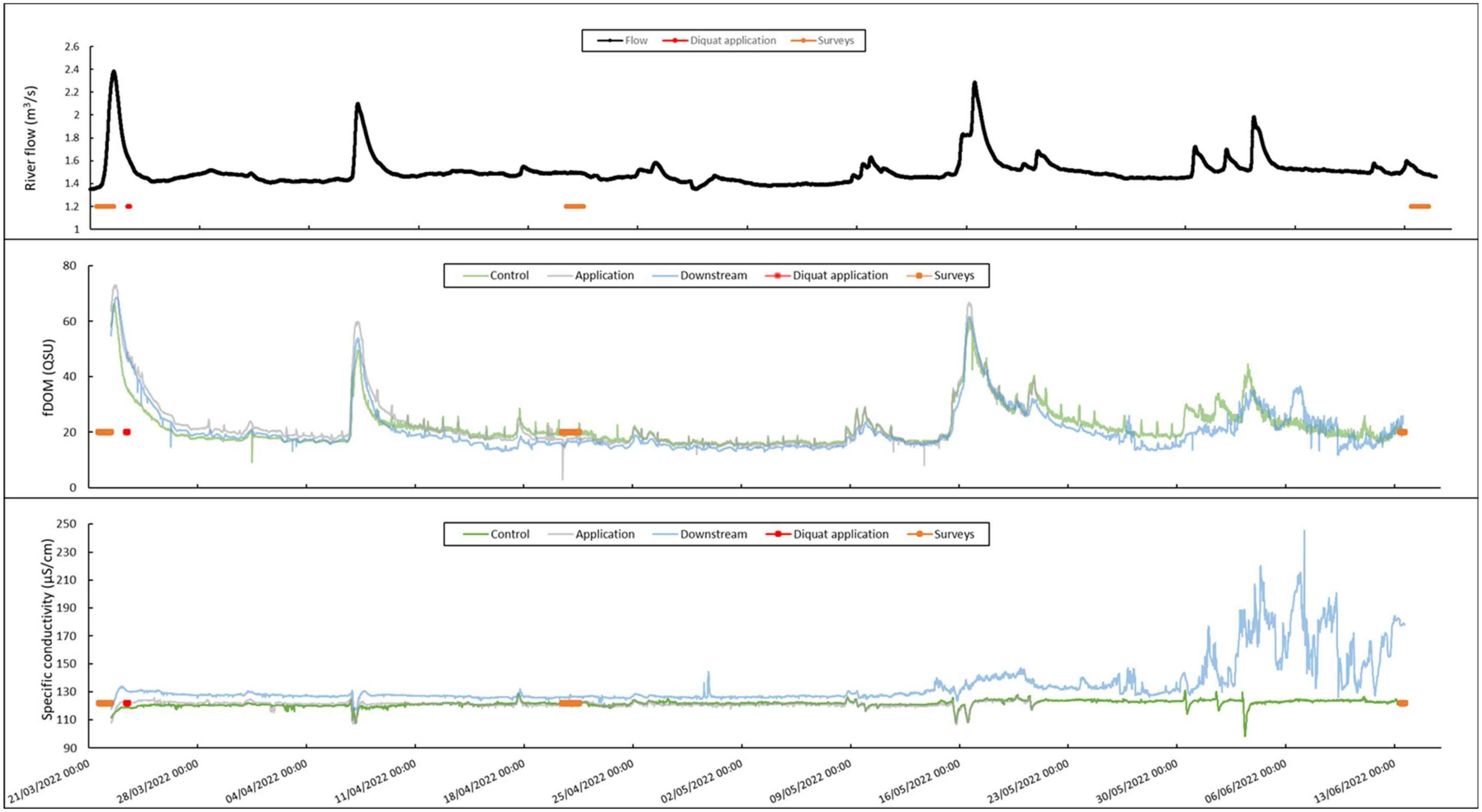

Figure 3-12: Flow, fluorescent dissolved organic matter (fDOM) and specific conductivity in the river during the diquat trial. Flow record from Environment Canterbury Radcliffe Road monitoring station. Data are daily flow averages. fDOM and specific conductivity data were logged at 15 min intervals by sonde instruments deployed in the centre of each trial reach (control, application, downstream). fDOM is a surrogate for CDOM (coloured dissolved organic matter). Orange markers indicate when aquatic plant surveys were undertaken.

Figure 3-13: Water depth, turbidity, and chlorophyll *a* in the river during the diquat trial. Data were logged at 15 min intervals by sonde instruments deployed in the centre of each trial reach (control, application, downstream). Orange markers indicate when aquatic plant surveys were undertaken.

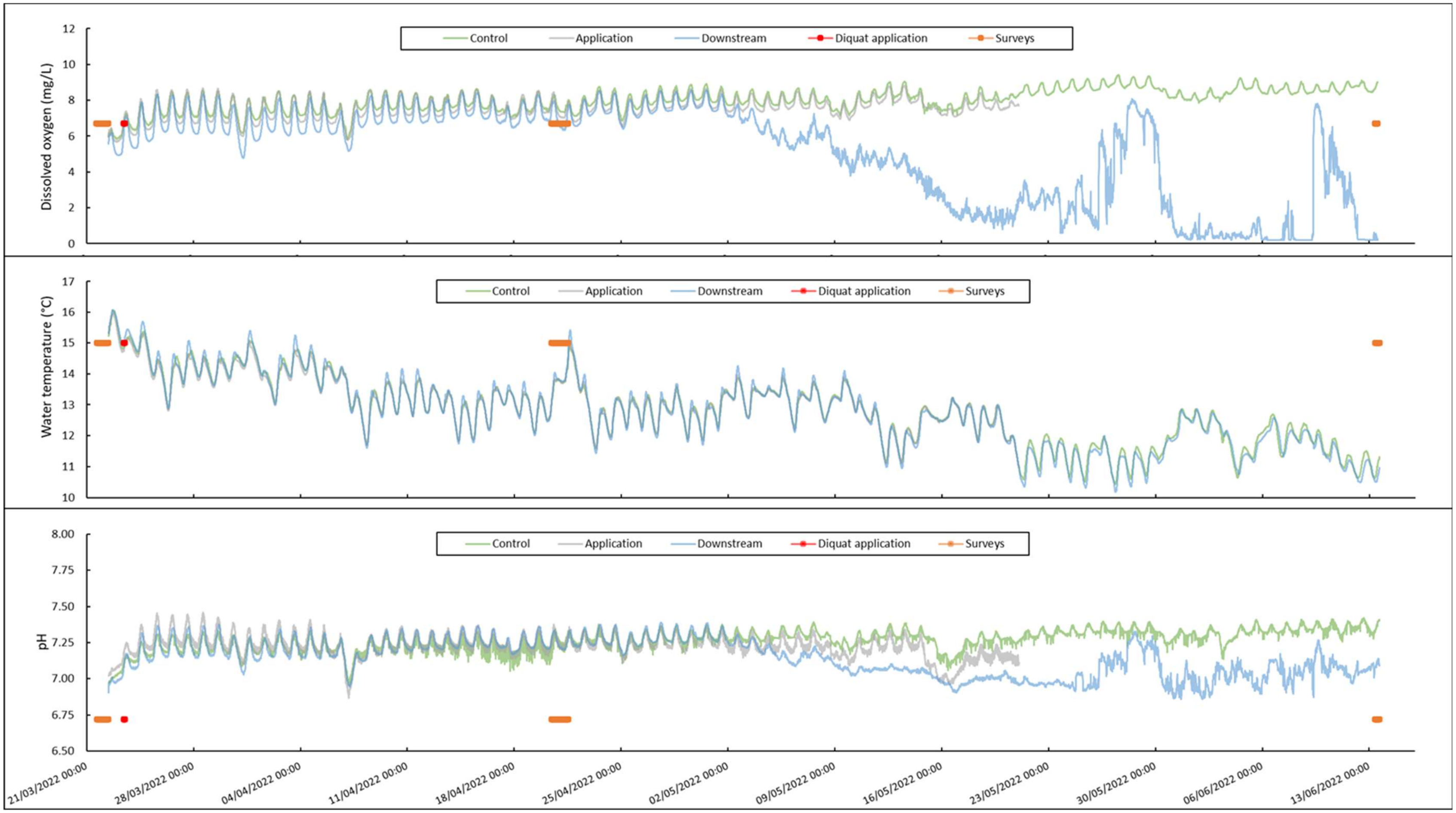

Figure 3-14: Dissolved oxygen, water temperature and pH in the river during the diquat trial. Data were logged at 15 min intervals by sonde instruments deployed in the centre of each trial reach (control, application, downstream). Orange markers indicate when aquatic plant surveys were undertaken.

3.4 Riparian shade evaluation

3.4.1 Fine-scale assessment

The average shade measured with a densiometer at the ten channel cross-sections ranged from 0% to 64%. Overall, and in each season, there was a significant inverse relationship between shade and aquatic plant clogginess at the ten survey sites (Figure 3-15, Figure D-1). Clogginess was inversely related to extent of shade. With no shade, clogginess averaged 33% while at sites with >60% shade clogginess was approximately halved, averaging 17%. At less shaded sites, clogginess showed a stronger tendency to increase from spring to summer to autumn, as indicated by the diverging seasonal regression lines. Clogginess in summer and autumn would have been reduced by weed harvesting activity.

For three of the four main submerged species present in the river, especially the two introduced weeds (i.e. elodea, curlyleaf pondweed), there was a noticeable decrease in clogginess with shade above 50% (Figure 3-16). In contrast, the native stonewort, *Nitella* sp. aff. *cristata*, recorded its highest clogginess (in range of 20-30%) at locations with mid-to-high shade.

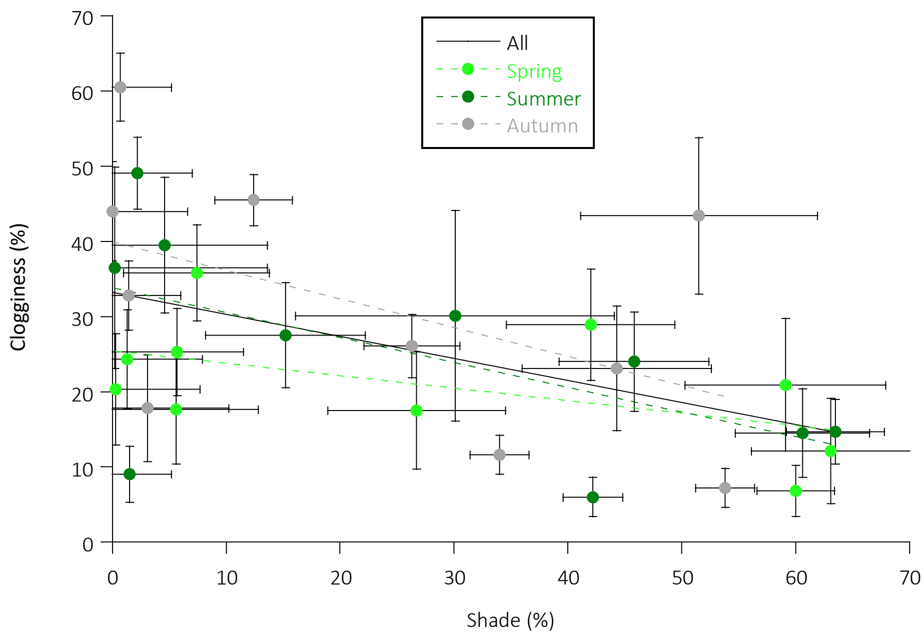

Figure 3-15: Relationship between macrophyte clogginess and shade. Shade measured with a densiometer. Clogginess and shade values are cross-sectional means (± standard error shown as vertical and horizontal bars, n=5). A plot showing the point estimate data and associated regression lines is provided in Appendix C. Black line and equation are for all data across the three survey dates. The relationship is statistically significant ($y = -0.2926x + 33.2$, $r^2 = 0.2539$, p=0.004**). Seasonal regression lines are the lighter dashed lines shown.

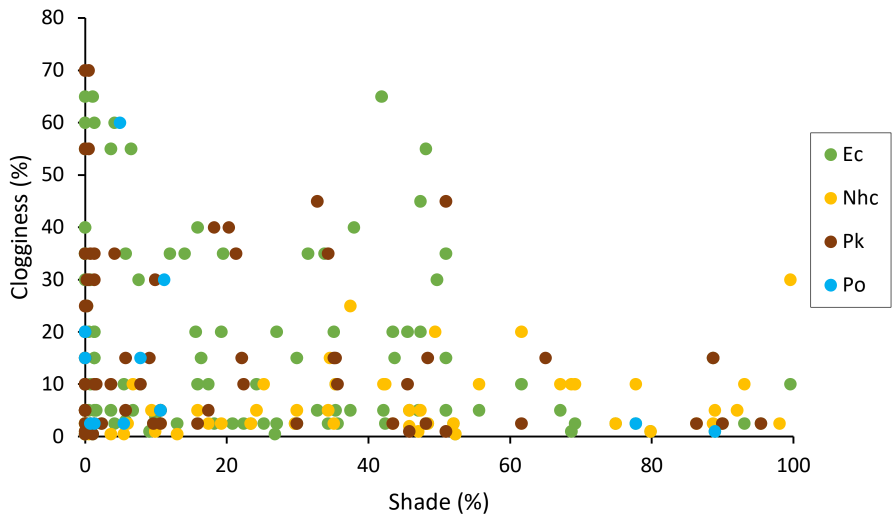

Figure 3-16: Clogginess records for the four main aquatic plant species associated with shade. Pk = *Potamogeton crispus* or curlyleaf pondweed, Ec = *Elodea canadensis* or elodea, Nhc = *Nitella* sp. aff. *cristata* or stonewort, Po = *Potamogeton ochreatus*, or blunt pondweed. 267 records shown for all points in the ten surveyed river cross-sections with aquatic plants in spring, summer and autumn.

Shade was also assessed in the channel centre (50% of wetted width) using a clinometer. This assessment yielded broadly similar values to the densiometer, although with considerable data scatter (Figure D-2). Shade was also modelled for the cross-sections from LiDAR data as described in section 2.3. Despite appreciable data scatter, modelled shade values were consistently higher than those measured by densiometer and clinometer, on average by 38 and 41% respectively (Figure D-3, Figure D-4).

Riparian vegetation at the cross-section sites was predominantly deciduous vegetation (frequently willows) and/or rank grasses (Figure 3-17). Native trees and shrubs in the riparian zone were generally more common at sites upstream of Dunlop Road.

With a predominance of deciduous riparian vegetation, a decrease in canopy cover, and therefore shade, was expected from summer to autumn as trees began to lose their leaves. However, only at Site 1 was a significant reduction in shade between summer and autumn observed (Figure 3-18). At Sites 3, 6 and 8 shade decreased from spring to autumn, with the largest reduction in shade (from 60 to 35%), at Site 6 at Turners Loop. This site was surrounded by tall deciduous trees, mainly willow.

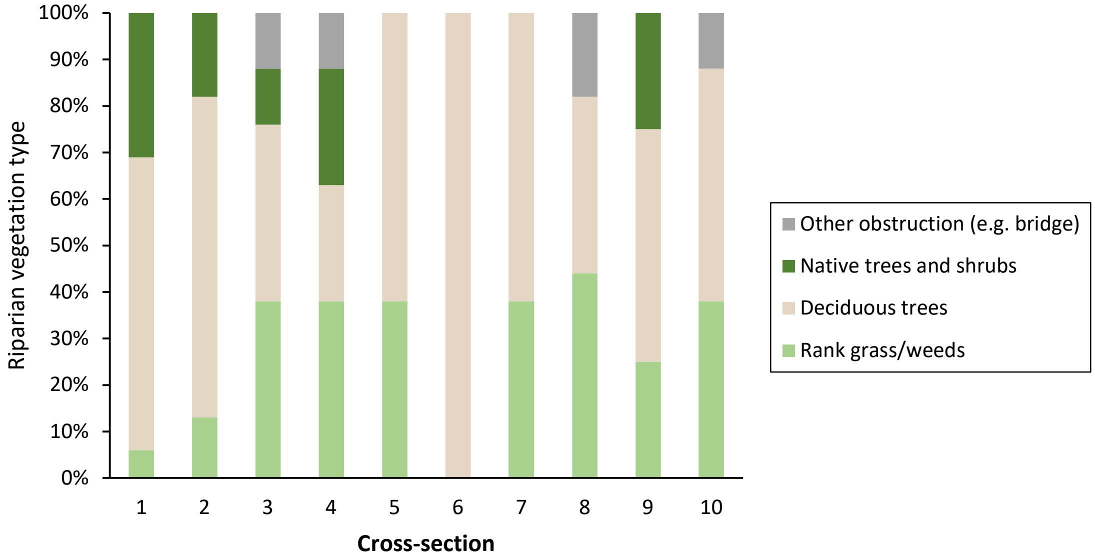

Figure 3-17: Riparian vegetation type at cross-section survey sites. Figure shows percent of each riparian vegetation type on banks based on clinometer survey data. Also includes other structures that create shade. Site 1 = Redwood springs, 2 = Willowview Drive, 3 = Radcliffe Rd, 4 = Janet Stewart Reserve, 5 = Dunlop Homestead, 6 = Turners Loop, 7 = Lower Styx Rd, 8 = Spencerville Bridge, 9 = Boat Ramp Reserve, 10 = Kainga/Harbour Rd.

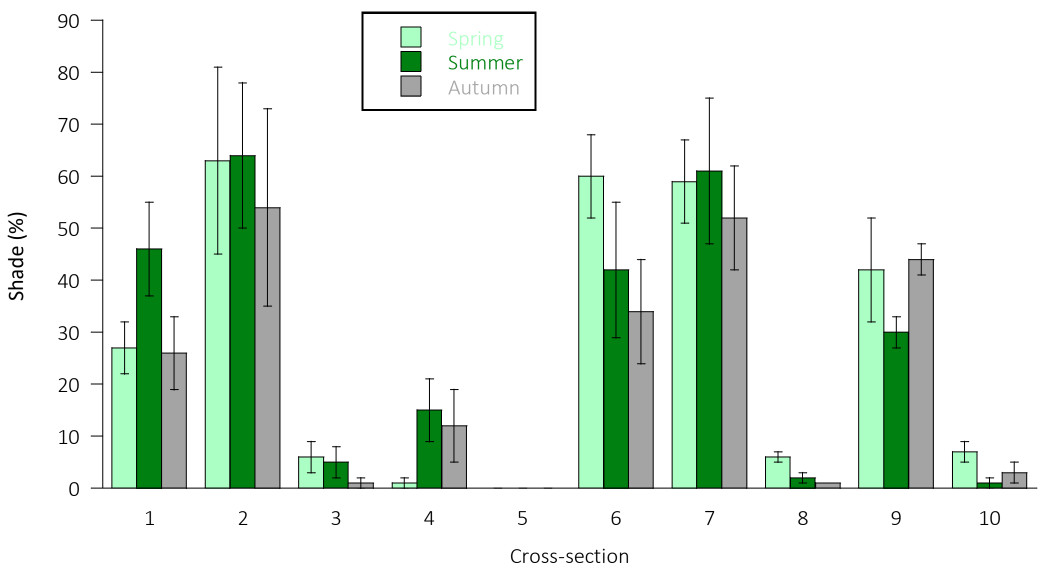

Figure 3-18: Shade at survey cross-sections by season. Columns indicate mean values. Bars indicate standard error (n=5). Data from densiometer observations. Site 1 = Redwood springs, 2 = Willowview Drive, 3 = Radcliffe Rd, 4 = Janet Stewart Reserve, 5 = Dunlop Rd, 6 = Turners Loop, 7 = Lower Styx Rd, 8 = Spencerville Bridge, 9 = Boat Ramp Reserve, 10 = Kainga/Harbour Rd.

3.4.2 Broad-scale assessment

Analysis of 146 transects using LiDAR and ArcGIS software tools indicated that the riparian vegetation along the banks of the river within 5 m distance of the channel was up to 35 m in height (Figure 3-19). River wetted widths ranged from 4 to 28 m. Modelled shade values ranged from 14 to 91% (average: 55%).

To achieve average shading of 70% for nuisance weed control (Matheson et al. 2018), a simple rule of thumb suggests that a 1:1 ratio of vegetation height to river wetted width is required (Davies-Colley and Rutherford 2005, Hawkes Bay Regional Council, NIWA, DairyNZ 2020). Ideally, any planting setback and bank height would also be included in the determination of this ratio, but these parameters were difficult to account for. The effects of these parameters are considered minimal and compensating, because an increase in bank height increases shade while an increase in setback *reduces* shade., We therefore chose to neglect their influence in this assessment.

Between transects 1 and 45 (Redwood Springs to just upstream of Site 5, Dunlop Road), the 1:1 ratio of vegetation height to wetted width was often achieved (Figure 3-19). However, this ratio was rarely met downstream of transect 45 (or cross-section Site 5, Dunlop Road), due to a combination of increased wetted width and decreased vegetation height. Nevertheless, in general the addition of around 10 m of extra vegetation height (to one bank or shared between both banks) would be sufficient to achieve the 1:1 ratio at open-canopy locations.

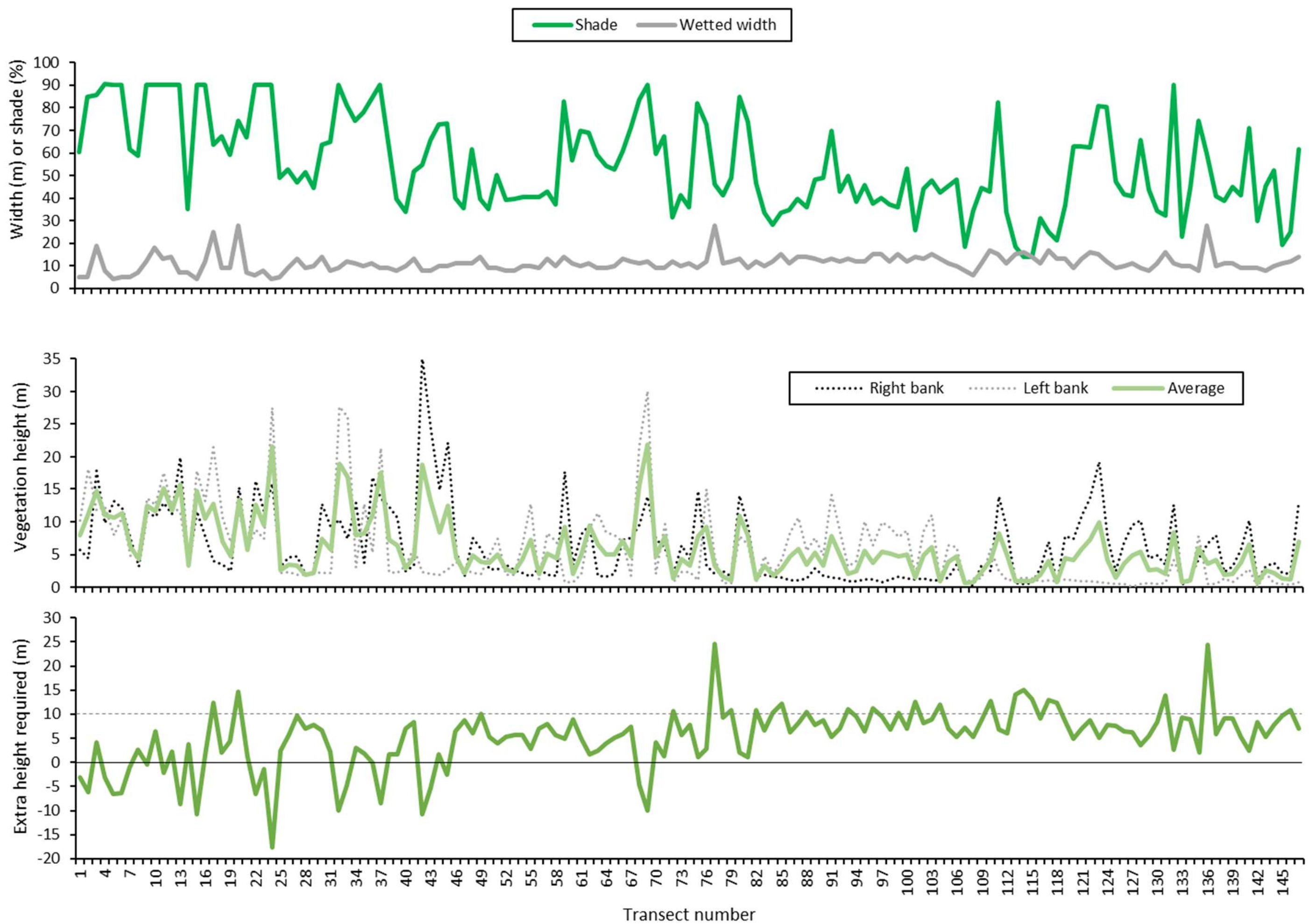

Figure 3-19: Modelled shade, wetted width and riparian vegetation height from LiDAR data. Model predictions made for transects at 100 m intervals. Analysis included determining extra height of riparian vegetation required to achieve 1:1 ratio of vegetation height to wetted width for 70% shade to achieve nuisance weed control.

3.5 Water quality, sediment, and climate effects

3.5.1 Water quality trends

This analysis aimed to identify trends in water quality variables that may explain the anecdotal perception of increased weed growth.

Monthly water quality data from the three Pūharakekenui/Styx River monitoring sites for the period mid-2007 to mid-2019 were analysed with flow adjustment applied (see Appendix F for trends and data with and without flow adjustment). The upper site was at Marshland Road, the middle site at Richards Bridge and the lower site at Kainga/Harbour Road. The following temporal trends were identified:

- A very likely decreasing trend in *conductivity* at the upper site, and likely decreasing trends at the middle and lower sites.
- Extremely likely decreasing trends in *dissolved inorganic nitrogen* (*DIN*) at all sites.
- Virtually certain decreasing trends in *dissolved reactive phosphorus* (*DRP*) at the upper and middle sites, and an extremely likely decreasing trend at the lower site.
- Likely decreasing trends in *nitrate-N* at upper and middle sites, and a very likely decreasing trend at the lower site.
- Extremely likely decreasing trends in *salinity* at upper and middle sites, and a likely decreasing trend at the lower site.
- A likely decreasing trend in *total-N* at the upper site, a virtually certain decreasing trend at the middle site, and an extremely likely decreasing trend at the lower site.
- A likely increasing trend in *total-P* at the upper site, an extremely likely increasing trend at the middle site, and a very likely increasing trend at the lower site.
- An extremely likely increasing trend in *total suspended solids* (*TSS*) at the middle site, and a likely increasing trend at the lower site.
- Virtually certain increasing trends in *water temperature* at all sites (Figure 3-20).

3.5.2 Climate and hydrological trends

Climate and hydrological data were examined with a similar objective. Climate data were obtained from the nearest monitoring station at Christchurch Airport. The analysis showed that from mid-2007 to mid-2019 there was a virtually certain increasing trend in maximum daily air temperature, and a very likely increasing trend in minimum daily air temperature, both consistent with global warming reports (Figure 3-20). Daily radiation also exhibited an extremely likely increasing trend over this period while daily sunshine hours showed no trend (see Appendix F, Figure F-3).

Trend analysis showed a virtually certain increasing trend in daily baseflow at this site from mid-2007 to mid-2019, and a very likely increasing trend in daily total flow over the same period (Figure 3-21). An increase in baseflow of c. 0.16 m^3/s is indicated for this 12-year period. At all hydrological monitoring sites on the mainstem of the river (i.e., Radcliffe Rd, Lower Styx Rd, Harbour Rd) a virtually certain increasing trend in water level was evident from the analysis (Figure 3-22). At Radcliffe Rd, a c. 0.23 m rise in level is indicated for the 12-year period.

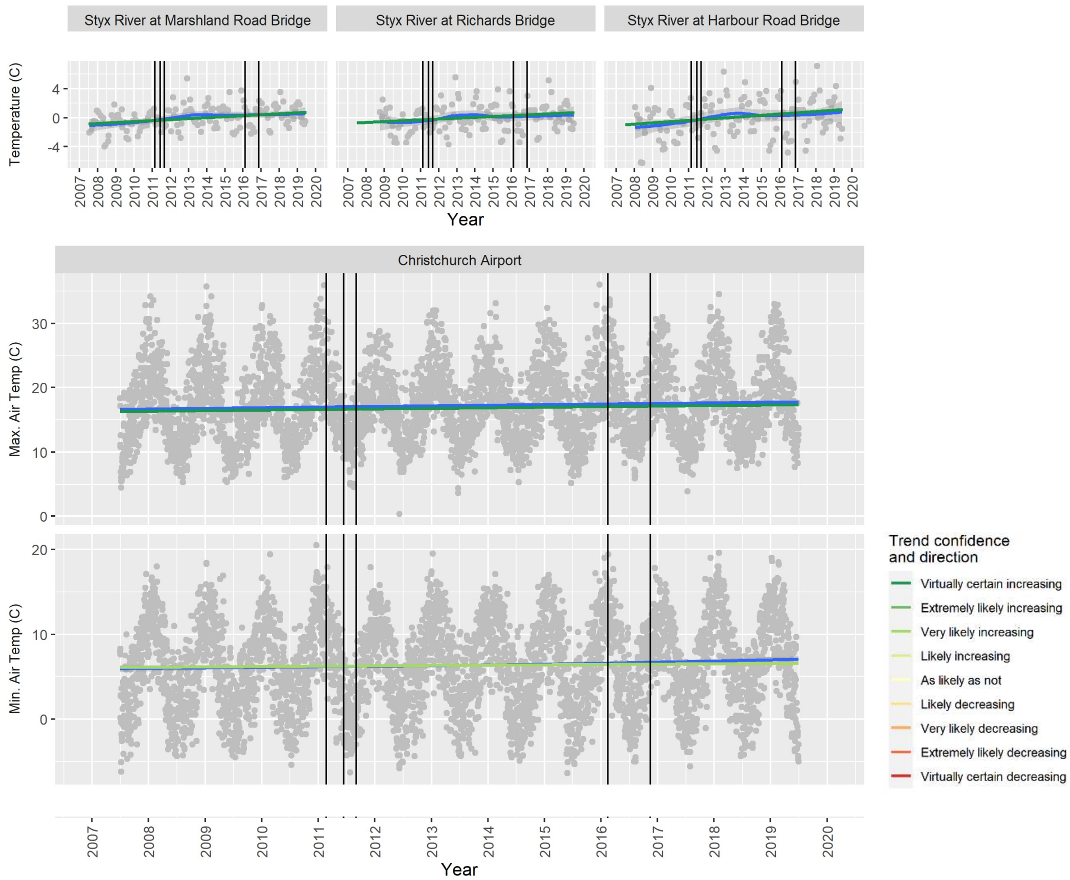

Figure 3-20: Water and air temperature trends. Top: flow-adjusted water temperature data for river mainstem monitoring sites are shown. Bottom: Air temperature data from Christchurch Airport. See Appendix F for all water quality data and trends with and without flow adjustment, and for other climate data and trends. Water quality sites ordered left to right from upper to middle to lower river. Vertical lines show significant earthquake occurrences from 2011 (4 September 2010 earthquake not shown). Blue line shows a flexible local polynomial regression (LOESS) for water temperature or generalized additive model (GAM) for air temperature, curve fit to the data for comparison with the monotonic trend detection line. Water temperature and maximum air temperature trends are virtually certain increasing. Minimum air temperature trend is very likely increasing. Rates of water temperature increase were 0.14, 0.11, 0.17°C/yr from upper to lower river sites. Rates of increase in air temperature were 0.04 and 0.09°C/yr for minimum and maximum records, respectively .

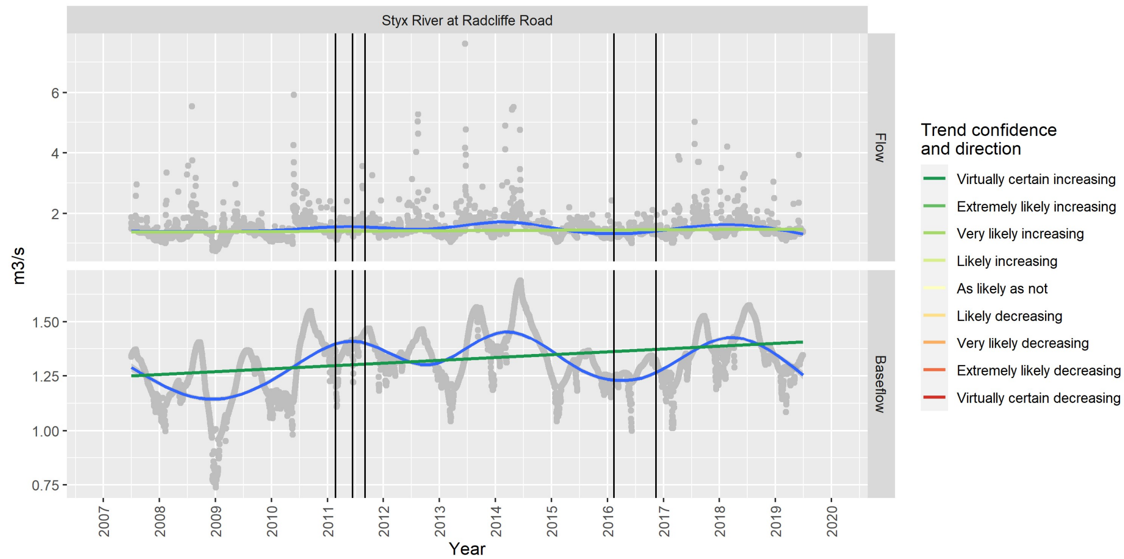

Figure 3-21: Trend analysis of Radcliffe Road flow data. Top: daily flow. Bottom: daily baseflow. Vertical lines show significant earthquake occurrences from 2011 (4 September 2010 earthquake not shown). Blue line shows a flexible Generalized Additive Model (GAM) curve fit to the data for comparison with the monotonic trend detection line. Flow trend is very likely increasing, baseflow trend is virtually certain increasing. Annual rates of increase were 0.008 and 0.013 m^3/s for flow and baseflow, respectively.

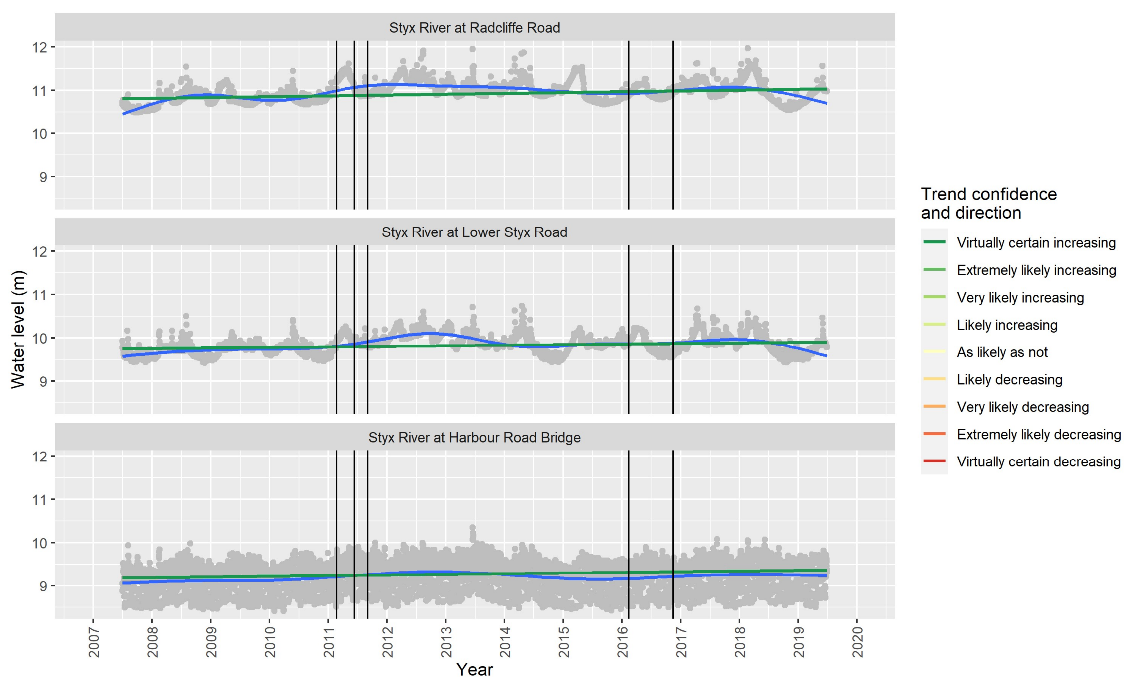

Figure 3-22: Trend analysis of River Styx water level data. From top to bottom: Radcliffe Rd, Lower Styx Rd, Harbour Rd. Vertical lines show significant earthquake occurrences from 2011 (4 September 2010 earthquake not shown). Blue line shows a flexible Generalized Additive Model (GAM) curve fit to the data for comparison with the monotonic trend detection line. All trends are virtually certain increasing. Rates of water level increase were 0.019, 0.014, 0.015 m/yr from top to bottom, respectively.

The days of accrual following a high-flow event that was twice, three or four times the median flow was also examined because these events can potentially re-set weed biomass in the river. Flow data from the Radcliffe Road Site was used to determine whether flow stability in the river may have increased over time. However, there was no evidence from the data that this has occurred (See Appendix E, Figure F-4). Flow events of three and four times the median flow were considered rare with only eight and two occurrences, respectively, over the studied period. Flow events that were twice the median flow were more common, often occurring several times per year. However, the record indicates a long period of flow stability without any high-flow events (≥2 times the median flow) between mid-2014 and early-2017.

3.5.3 Water nutrients

Water samples collected at each of the ten cross-sections during the three seasonal surveys showed that nitrate-N concentrations were generally in the range of 200 to 500 µg/L, with lowest concentrations at the most upstream and downstream sampling locations Sites 1 and 10, respectively, and highest concentrations at Sites 4 and 5 (Figure 3-23). Ammoniacal-N and dissolved reactive-P concentrations were generally less than 30 µg/L and showed a tendency to increase in a downstream direction. Overall nutrient concentrations showed a decreasing trend from spring to summer to autumn. Water turbidity and conductivity were generally low, including at the time our samples were collected.

3.5.4 Sediment nutrients

The total N and P contents of sediments in the river were variable (Figure 3-24). Average concentrations did not increase in a consistent manner, in a downstream direction, as might be expected. Average TN per transect ranged from 0.14 to 0.90% of sediment dry weight (DW), and average TP per transect ranged from 0.04 to 0.16% DW.

3.5.5 Nutrients and weed clogginess

Macrophyte biomass at survey sites, indicated by clogginess, was not correlated with measured concentrations of sediment or water nutrients (Figure 3-25). Sediment nitrogen was low and potentially at limiting concentrations in some locations (i.e., <0.1% DW, see Appendix E, Table E-3) but dissolved inorganic nitrogen concentrations in the water were at levels considered adequate to support growth (>100 µg/L). Sediment phosphorus was likely non-limiting (i.e., >0.01% DW) but at several locations dissolved reactive phosphorus concentrations were low and at potentially limiting levels (<10 µg/L).

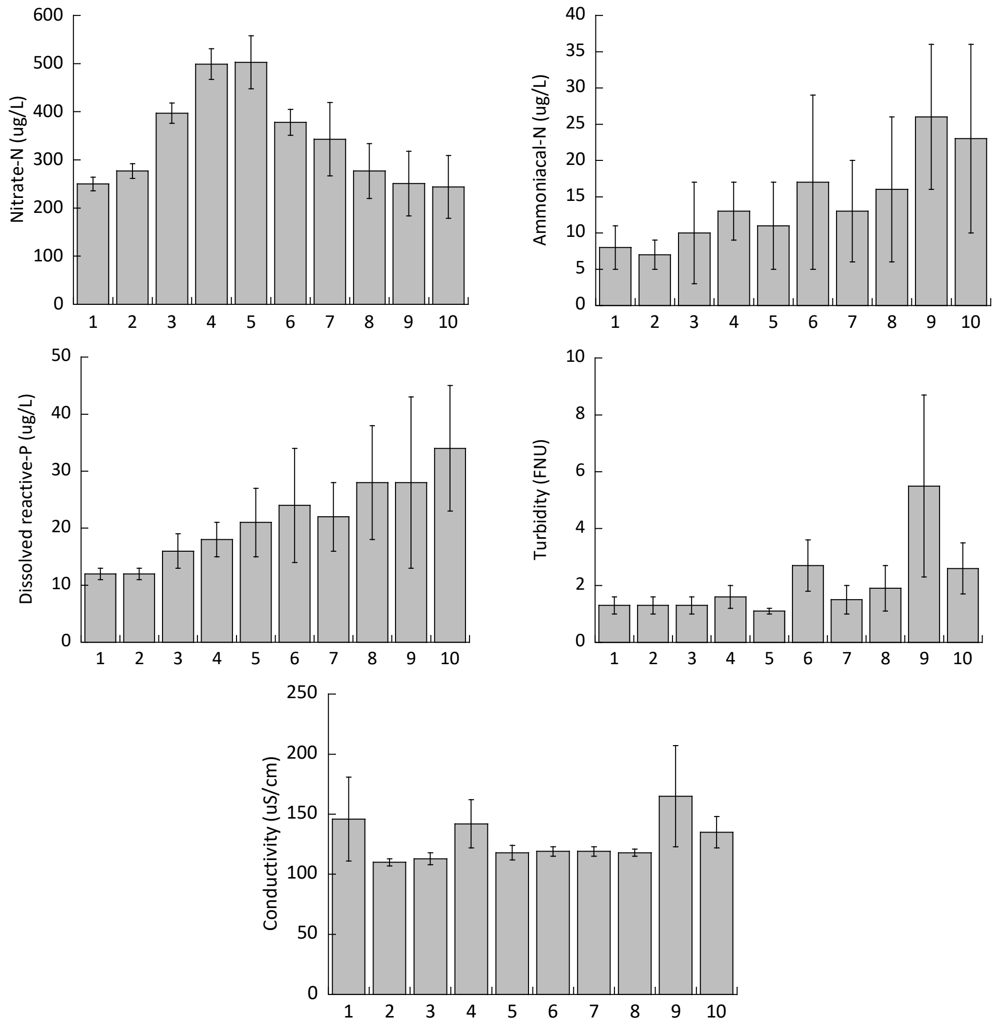

Figure 3-23: Water quality results for river cross-section sites sampled during seasonal surveys. Top left: Nitrate-N. Top right: Ammoniacal-N. Centre left: Dissolved reactive-P. Centre right: Turbidity. Bottom: Conductivity. Site numbers ascending from upstream to downstream. Values are means (± standard error) of spring, summer, and autumn samples.

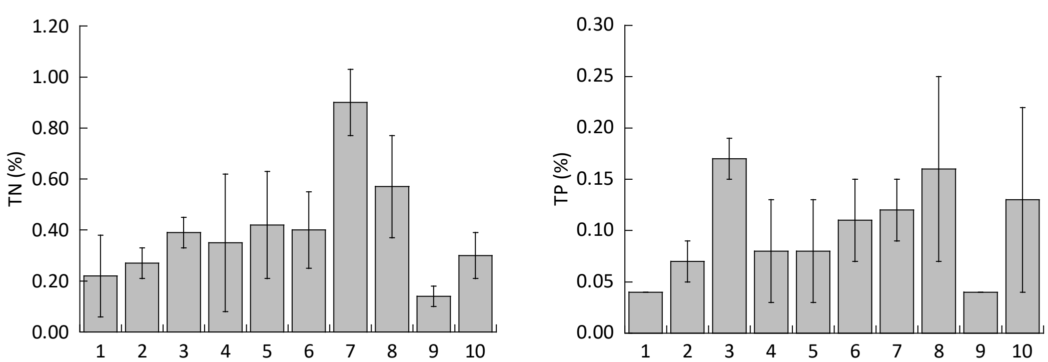

Figure 3-24: Nitrogen and phosphorus contents of surficial sediments at cross-section sites. Samples (0-2 cm depth) collected 9 November 2020 at 10, 50, 90% of cross-section wetted width at each site. Values are mean (±standard error, n=3).

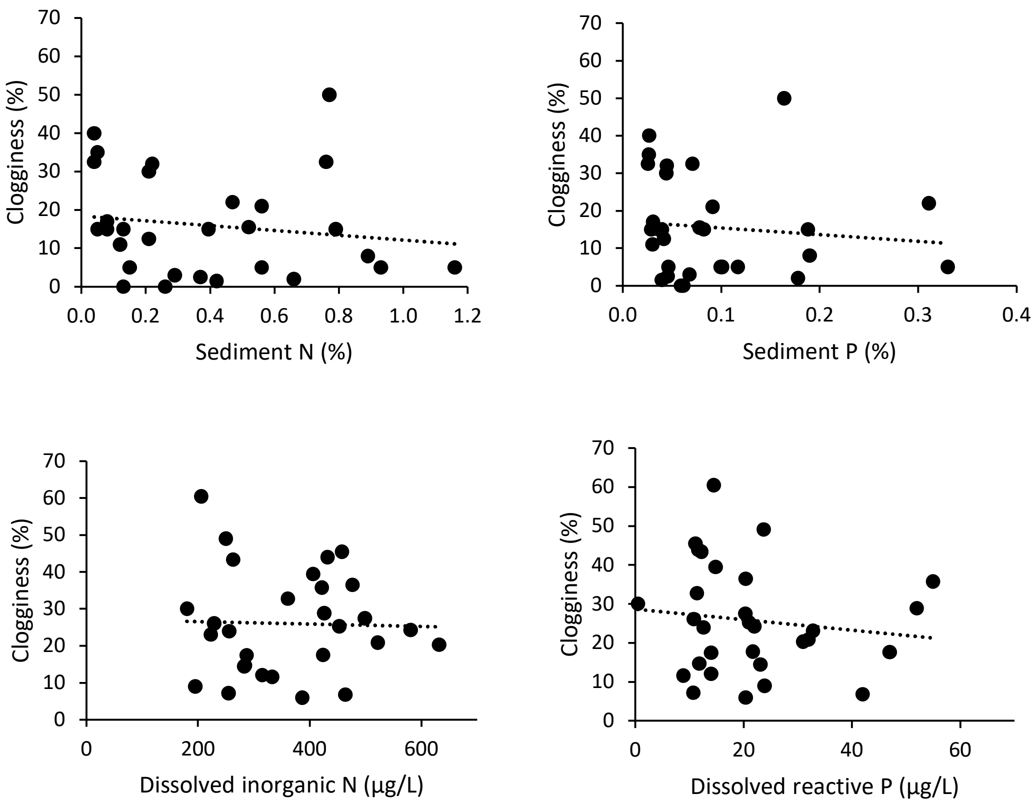

Figure 3-25: Sediment and water nutrients related to macrophyte clogginess. Sediment data collected in November 2020 from the ten channel cross-sections, at points at 10, 50 and 90% of wetted width (n=30). Corresponding clogginess data for these points are shown. Water nutrient data collected in November 2020, February 2021 and April 2021 in channel centre at each cross-section (n=30). Dissolved inorganic nitrogen is NO_3-N + NH_4-N. Linear regression lines are shown but relationships are not statistically significant.

4 Discussion

4.1 Delimitation of weed nuisance

The weed delimitation was carried out to better understand the current state of the weed problem in the river to inform future management.

The dominant aquatic weeds in the Pūharakekenui/Styx River from Redwood Springs downstream were the two introduced species, elodea and curlyleaf pondweed. These two species contribute most of the weed "clogginess" in the river. Clogginess of the channel was high (>50%) in some locations but averaged 17 to 30% based on the broad-scale hydroacoustic surveys and fine-scale assessments conducted during the 2020-21 growing season. Elodea was slightly more abundant than curlyleaf pondweed in all seasons (spring, summer, and autumn) and tended to grow in deeper water.

Two native species were also conspicuous in the river, although their abundance was generally much less than that of the two introduced species. The stonewort, *Nitella* sp. aff. *cristata*, was relatively common throughout, while blunt pondweed (*Potamogeton ochreatus*) only occurred in the lower part of the river. The relative abundance of species is considered to be affected by the harvesting regime (Taylor et al. 2000), with the two introduced species better adapted to physical disturbance (Riis and Biggs 2001). However, these two species would likely outcompete the native macrophytes, even in the absence of harvesting disturbance due to faster growth rates and lateral spread (Riis and Biggs 2001). The generally taller and denser growing habit of the two introduced species is also more likely to impede flow.

Aquatic plant "clogginess" was greatest in autumn, despite weed harvesting removing an estimated 140 tonnes of weed in the summer. The autumn surveys indicated an average 30% clogginess of weed in the river at this time. Presumably, without the harvesting, the autumn weed clogginess may have been even greater. However, at certain sites weed beds were dense and may have been approaching maximum carrying capacity.

In the nearby Avon River, a more noxious weed, egeria (*Egeria densa*), is present and this poses risk of transfer and infestation of the Pūharakekenui/River Styx (Figure 4-1). Egeria has been present in the Avon River since 1998 (Wells and Sutherland 2001). It is an unwanted organism, listed on the National Pest Plant Accord and is actively managed outside of its core area of distribution in the North Island. Egeria is subject to a Progressive Containment Programme in Canterbury (Champion et al. 2019). It was inadvertently introduced to the Avon River by a weed harvester operating in both the Opawa River in Marlborough and the Avon River (Champion et al. 2019). The harvester that operates in the Pūharakekenui/Styx River also operates in the Avon River (Figure 4-1).

Egeria is a more aggressive weed species than either elodea or curlyleaf pondweed (Champion and Clayton 2000), and an infestation in the Pūharakekenui/River Styx may result in increased weed growth, biomass and "clogginess", and further displacement of native species. A recent New Zealand study has shown that egeria accrues biomass more quickly than elodea, especially where there is some shade, although shade at higher levels can constrain the growth of both species (Ellawala Kankanamge et al. 2019, 2020).

A low frequency of flushing flows also encourages egeria dominance, conditions that are prevalent in the Pūharakekenui/Styx River. In similar-sized rivers in the Hauraki Plains, where egeria, hornwort and curlyleaf pondweed dominate, Matheson and Wells (2017) confirmed that higher levels of aquatic plant clogginess are associated with long periods of stable flow. Clogginess was frequently correlated to the number of days of accrual following a flushing flow event equivalent to three times the median flow (DAFRE3), and clogginess >40% was associated with DAFRE3 of 80 days or more. Flushing flows of this magnitude are relatively infrequent in the Pūharakekenui/Styx River, occurring only eight times between mid-2007 and mid-2019, on average once every 460 days (min: 41 days, max: 1136 days) (see Figure F-4).

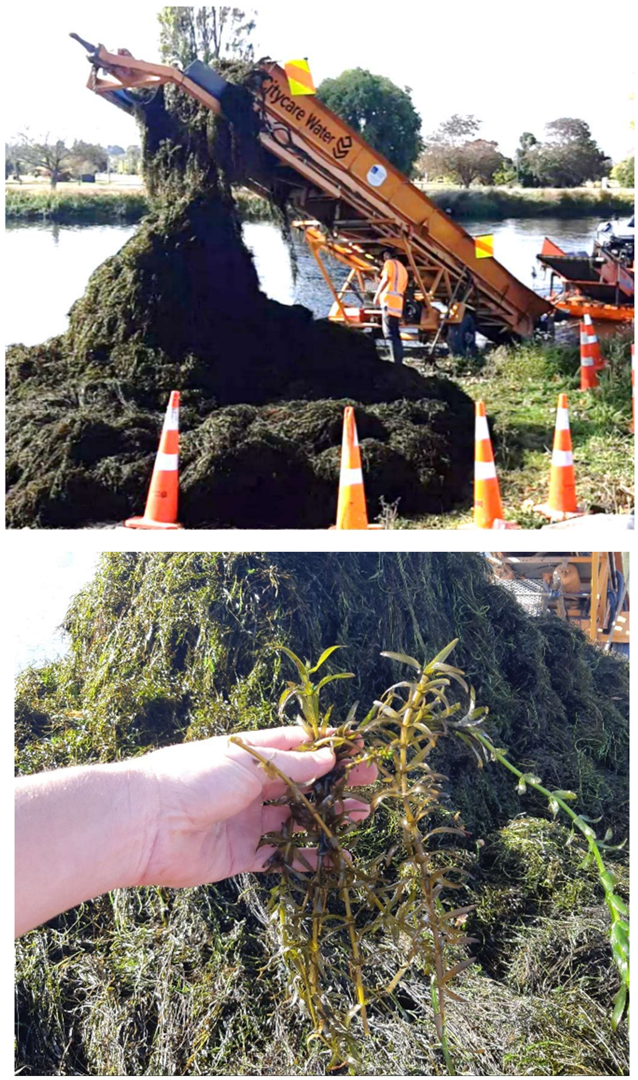

Figure 4-1: Weed harvesting in the Avon River – April 2021. Left: Harvested weed being offloaded to shore. Right: Egeria in amongst the harvested stockpile.

4.2 Harvesting review

A review of available data for the harvesting operation was undertaken to identify opportunities for improvement.

Analysis of these data suggests that harvesting generally reduces water level more when carried out later in the growing season. Aquatic weed biomass usually peaks towards the end of summer, early autumn or sometimes even later, especially if there is little physical disturbance (e.g., few flushing flows) (Champion and Tanner 2000, Matheson and Wells 2017). That late-season harvesting is more efficacious presumably reflects removal of a greater mass of accumulated weed at this time, which would likely have a stronger impact on flow conveyance than removal of the lesser biomass present in the early season period.

The long-term data suggest:

- a tendency over time for harvesting to increasingly decrease water levels in the lower river (increasing efficacy), whereas,
- a tendency over time for harvesting to increasingly have a smaller effect on water level in the upper river (decreasing efficacy).

However, data for the 2020-2021 season showed water level reductions of similar magnitude in both locations. It may be easier to extract weed in the deeper waters of the lower river – recent observations indicate that just over 70% of the harvested weed was removed from downstream of Lower Styx Rd in summer 2021.

The hydroacoustic survey team noted an extensive area of shallow water depth downstream of the Spencerville Road bridge, which potentially acts as a natural control on water level. The affected area covered an approximately 1.3 km length of the river channel, had dense growth of weed, and contained a 0.3 km stretch that was especially shallow (0.5-0.7 m water depth) (Figure 4-2). This area presumably restricts flow conveyance (and causes high water levels) in this segment of the river and may warrant further investigation.

Figure 4-2: Location of shallow flow constricted area downstream of Spencerville Road. Information based on the April 2021 hydroacoustics survey. In the c. 1.3km length of river between the two orange markers the channel is constricted, with dense curlyleaf pondweed in the shallows and elodea in deeper water. In the c. 0.3 km section between the red markers water depth was c. 0.5 – 0.7 m across the channel.

The harvester currently operates in both the Avon and Pūharakekenui/Styx Rivers. Transfer of egeria fragments (propagules) from the Avon poses a significant threat to the Pūharakekenui/Styx River. CityCare decontaminate the harvester when moving between waterways by spraying the equipment with the herbicide, glyphosate. Glyphosate is used to control emergent vegetation and is not used to control submerged weeds, like egeria. To decontaminate the harvester, we recommend following the Ministry for Primary Industries "check, clean and dry" protocols for boats: see Check, Clean, Dry: preventing didymo and other pests | MPI | NZ Government. It must be noted however that weed harvesters are a special, high risk case with increased potential for harbouring weed propagules, and extra care and attention should be taken to ensure that they are weed-free.

Internationally, decontamination using hot water is considered one of the most efficient, environmentally sound, and cost-effective methods to prevent the accidental spread of invasive species (Beyer et al. 2011, Stebbing et al. 2011). In previous NIWA trials egeria exposed to 45°C water for 20 minutes had a 3% survival rate. No plants survived 55°C water exposed for 20 min or 60°C for 1 minute (Burton 2017). The use of hot water (>60°C) for decontamination of the harvester should be investigated for practicality. Steam has also been suggested as a potential control option for other aquatic weeds (van Oosterhout 2007) and might warrant evaluation for decontamination of the

Pūharakekenui/Styx River weed harvester (see the Alberti industrial medi-vap as an example of equipment that might be used).

4.3 Diquat trial

The small-scale trial (200 m application reach) for diquat resulted in a small effect on weed abundance over three months, with total weed clogginess reduced by c. 7% attributable to the herbicide in the application reach compared to the control and downstream reaches. In the month following diquat application total weed clogginess in the application reach reduced from 55% to 37%, and to 18% over three months, but this could not be distinguished from a seasonal reduction in clogginess. Statistical analysis of the survey data did not identify a significant effect of diquat on clogginess of aquatic plants, total and by species. The native aquatic plants, milfoil and stonewort were unaffected by the application, and in fact, increased slightly in abundance in the month following diquat application.

Water sampling did not detect any diquat residues (≥0.001 mg/L) in the river after its application, and no effects on river water quality, including dissolved oxygen, were detected by continuous monitoring using sondes.

Diquat application in the nearby Avon River was shown to be highly effective at controlling elodea (and egeria), and it also reduced the abundance of pondweeds (curlyleaf and blunt) by c. 30% (Wells and Sutherland 2001). In experimental trials, elodea and curlyleaf pondweed have shown considerable sensitivity to diquat with complete degradation of shoots 2.5 weeks after emersion in a solution containing 1 mg/L diquat for 10 and 30 mins, respectively (NIWA, unpublished data). The reduced efficacy in the present trial was unexpected and may be attributed to several factors.

The timing of the diquat application in the present trial unfortunately coincided with rain in the preceding 48-hour period, and an elevated river level. Flow records for Environment Canterbury's Radcliffe Road monitoring station show that river discharge at this location increased by 57% from 1.36 to 2.38 $m^3 s^{-1}$ in the 48-hour period preceding diquat application. The increased discharge, flow velocity and water depth in the application reach would have reduced the likelihood of achieving good contact between the herbicide once applied and the target weed beds. The Reglone label recommends that diquat is not applied when flow velocity exceeds 0.3 m/s. No data is available on flow velocity at the time of diquat application, but a maximum flow velocity approaching this, of 0.26 m/s was measured at this location in February 2022 (NIWA unpublished data). A better result might be expected under stable flow conditions so repeating the trial under these conditions is recommended.

It is also possible that the diquat applied was rapidly adsorbed by dissolved organic matter or particles in the water, or detritus on the weeds, reducing its efficacy (Clayton and Matheson 2010). Weed condition, however, was checked by the applicator and considered to be at a level that was satisfactory (Boffa Miskell 2022).

A spring diquat application was initially recommended for this trial. However, the delimitation survey and harvesting review indicated autumn as a critical time for weed accrual in the river, so an autumn trial was approved. Spring and autumn are the months usually recommended for diquat application, as rapid plant decay in warm summer months risks deoxygenation of treated waterbodies under low flow conditions, although we note that a January application was carried out in the Avon River (Wells and Sutherland 2001).

To evaluate further the suitability of diquat as a method to manage weed nuisance in the Pūharakekenui/Styx River we recommend testing the efficacy of diquat applied in spring versus early autumn (under stable flow conditions). Based on the results of the present trial a subsequent trial would benefit from including additional, higher resolution measurements of diquat. These measurements should be made within, upstream, and downstream of, the application reach during and immediately after application to improve understanding of the concentrations achieved, when and where, and enabling calculation of rates of dispersion/adsorption.

4.4 Riparian shade evaluation

Riparian shade in the river system was assessed to determine current state, its potential to regulate growth of the dominant weed species, and the potential for new riparian plantings to contribute to weed control.

Current state was assessed using fine-scale and broad-scale approaches. The fine-scale approach involved measurements of shade at ten cross-sections using a densiometer and a clinometer. The broad-scale approach used LiDAR data and the ArcGIS radiation tool combined with a semi-empirical approach to model shade at the ten surveyed cross-sections, as well as at 100 m intervals along the river from Redwood Springs to Kainga/Harbour Road. Our assessment found that measured average shade at the ten cross-sections ranged from 0 to 65%, while modelled average shade was consistently higher at 14 to 91%.

The discrepancy between measured and modelled shade is most likely a consequence of overestimation by the model algorithm. For cross-sections and transects along the river where riparian canopy overhang occurred, the ArcGIS radiation tool could not be used, so in this situation shading of 90% was assumed. This occurred relatively frequently (21% of 735 point estimates). Actual shading in these cases may be appreciably less than 90%, especially if the density of the vegetation canopy is low. This is highly likely given that willows dominate the riparian vegetation along the river. For example, at Turners Loop where tall, exotic riparian vegetation (mostly willows) dominated, average measured and modelled shade was 60 and 89%, respectively.

Measurements at the ten cross-sections showed a negative relationship between shade and weed clogginess, suggesting that riparian shade has potential for use as a management tool to suppress the growth of aquatic weeds. This is broadly consistent with the findings from other recent New Zealand studies (Matheson et al. 2018, Mouton et al. 2019, Ellawala Kankanamge et al. 2019, 2020), and consistent with previous recommendations for the Pūharakekenui/Styx River (McComb 1997, CCC 2003, van den Ende and Patridge 2008). It also provides new information for managers of rivers infested with these two weed species, elodea and curlyleaf pondweed. Results from the Pūharakekenui/Styx River indicate that a gradual trend of decreased weed abundance with increased shade exists (rather than an abrupt threshold), at least for the dominant weed species present in this river.

LiDAR data and ArcGIS software were used to assess the potential for new riparian plantings to increase shade and reduce submerged weed abundance. When riparian vegetation height was greater than or equal to channel wetted width (i.e., ratio ≥1), shade ≥70% was assumed based on Davies-Colley and Rutherford (2005). This level of shading is considered sufficient to prevent aquatic weed clogginess exceeding 50% (Matheson et al. 2018). Measurements from the Pūharakekenui/Styx River are consistent with this expectation because clogginess averaged only 17% with shading of >60%.

In locations where the ratio of vegetation height:wetted width was <1, the amount of extra vegetation height required to achieve a ratio of 1 was determined. Our analysis showed that in most parts of the river, planting of riparian vegetation to add an extra 10 m of height to one bank or split across both banks (e.g., extra 5 m each side), would be sufficient to achieve this ratio. From the results mentioned previously, even smaller riparian height additions (e.g., 1 to 3 m per bank) would likely reduce weed growth rates and clogginess. Ideally, additional riparian vegetation should be installed, as dense plantings of evergreen natives, are able to achieve high and consistent shading levels as well as contributing to indigenous biodiversity. We note that shading may increase the competitive advantage of the native stonewort to dominate and maintain lower levels of the introduced aquatic weeds, elodea and curlyleaf pondweed.

If planting is only practical on one bank, then where the river channel is oriented from west to east (or vice-versa) tall plantings on the north bank combined with low plantings on the south bank should still create sufficient shade (Rutherford et al. 2021).

4.5 Nutrient and climate effects

Nutrients (N and P), in Styx River water and sediment, are adequate to support growth of nuisance aquatic plants, and show decreasing trends. The recent anecdotal increase in nuisance weed growth in the river is therefore unlikely to be the result of an increased supply of nutrients.

The trends analysis showed trends of increasing water and air temperatures, and daily radiation over the 12-year monitoring period. The magnitude of observed trends were consistent with those attributed to global warming (MfE 2018). Aquatic plant growth rates are often enhanced by increased water temperatures and more radiation (Matheson et al. 2012, Riis et al. 2012), so this could plausibly explain an increase in weed nuisance. For example, growth rates for elodea have been shown to increase by a factor of 10 between 5 and 15°C, with 5°C being a lower limit for growth (Madsen and Brix 1997). Work by Riis et al. (2012) suggests that growth rates for elodea may peak at around 25°C as they found lower growth rates at 20 and 30°C. They also found that growth rates of egeria were higher than those of elodea with warmer temperatures. Curlyleaf pondweed is reputedly more of a cold-water strategist, being able to survive under ice at temperatures of 1-4°C. Nevertheless, like elodea, this species grows vigorously at temperatures between 5 and 20°C (Boulduan et al. 1994), consistent with the range of water temperatures generally encountered in the Pūharakekenui/Styx River (see Appendix F). However, over-wintering turion (detached bud) production in curlyleaf pondweed is reported to be maximal at water temperatures >15°C and day lengths >12 hr (Boulduan et al. 1994). Therefore, temperatures above this could reduce growth and induce senescence in mid- to late summer. Studies in North America show that this species naturally senesces in summer after turion production (Tobiesson and Snow 1984, Netherland et al. 2000). Senescence of this species was not evident from the Pūharakekenui/River Styx clogginess measurements, but there were obvious signs of chironomid browsing damage in autumn.

Increasing trends in water level suggest an increasing risk of flooding in low-lying areas of the catchment. Trends of increased flow and baseflow suggest that elevated water levels are the result of the increased discharge, probably exacerbated by restrictions to flow conveyance caused by weed and sediment accrual in the channel. However, it should be noted that obtaining accurate estimates of river flow in weed-infested waterways is difficult (Wilding et al. 2016, Bulleid 2019), and more regular checks and corrections to the stage-discharge relationship are recommended (NEMS 2016). Furthermore, flow is measured at only one location on the mainstem of the river at Radcliffe Road, so this finding may require some further investigation. The stronger increasing trend in baseflow (c.f. total flow) suggests that the increase is predominantly of groundwater origin.

Increases in baseflow might be due to earthquake-initiated changes, although no obvious increases in water level, total flow or baseflow around the times of the major recent (2010-2016) earthquakes in the Christchurch area are apparent from the assessed river hydrometric records. However, there is evidence from CCC LiDAR surveys that the 2010 earthquakes caused land settlement in the lower floodplain, and that there was an abrupt shift (increase) in Radcliffe Road water level of approximately 12 cm at this time (Harrington and Parson 2012).

5 Conclusions

Aquatic vegetation in the spring-fed, Pūharakekenui/Styx River north of Christchurch is currently dominated by the two exotic submerged weeds, elodea and curlyleaf pondweed. These species were also dominant when the river was last surveyed in 2008.

In 2020-21 abundance of these two weed species and other aquatic plants, including some native species, increased between spring and autumn survey dates until they occupied on average c. 30% of the river mainstem's wetted channel. This plant density characterised the channel downstream from the Radcliffe Road survey site. Mechanical harvesting removed c. 140 tonnes of weed in January-February 2021 and water levels dropped by up to 0.16-0.27 m at this time.

A small-scale application of the herbicide diquat in autumn the following year had a small to negligible effect on weed abundance in the reach to which it was applied, compared with upstream and downstream reaches. Elevated river flow conditions at the time of the application may have contributed to the low diquat treatment efficacy.

In areas where riparian vegetation shaded the water surface, aquatic plant clogginess was reduced by up to half, dependent on the level of riparian shading at any particular site. Planting additional riparian vegetation to create more shade has the potential to reduce nuisance weed growth in the river over time and favour native aquatic plants tolerant of higher shade (e.g., stonewort). To achieve this, the riparian planting should form a dense canopy along the river margin, with vegetation approximately as tall as the river is wide, ideally on both banks.

Increased weed growth in the river in recent years has been reported anecdotally and is considered a possible reason for recent increased water levels and flood-risk, especially in low-lying areas of the catchment, adjacent to the river. However, no long-term aquatic plant monitoring data are available to confirm that weed accrual has increased.

Increasing trend in water and air temperatures and solar radiation are plausible drivers of increased weed growth and biomass accrual. The 2020/21 river survey also revealed a shallow reach with high aquatic plant density in the lower river that is likely to impede flow in the lower river, and which could contribute to rising water levels. Analysis of the hydrometric data for the river suggests that river flows, especially baseflows, have increased over time. The combination of increasing baseflow, impeded drainage caused by extensive plant growth and possibly increased sediment accumulation (especially downstream of Spencerville Road) are likely to be the primary reasons for increased water levels and increased flood-risk.

Recommendations arising from this investigation are as follows:

- Identify and implement an appropriate decontamination method (e.g., 60°C water treatment for 1 min) as a compulsory Standard Operating Procedure for the harvester operator, to eliminate the risk of egeria introduction from the Avon River by this means. The use of a hot-steam high pressure gun may also be a practicable option.
- Undertake annual surveillance to maximise the potential to detect any early incursion of a more aggressive aquatic weed species, such as egeria, lagarosiphon or hornwort, and develop a response plan to ensure preparedness should an incursion occur. For convenience it may be possible to use a shore-based grab or rake sampler to carry out annual surveillance, but the most effective method of detecting a new incursion is

likely to be a snorkeling surveillance survey that focuses on areas around all key access points to the river.

- Train or provide guidance[6] to CCC and CityCare staff doing river monitoring and maintenance to enable them to identify these high-risk weed species and also provide signage to inform the general public.
- Investigate the removal of sediment accrued downstream of the Spencerville Road Bridge.
- Undertake further trials to evaluate the efficacy of diquat herbicide as an additional method for management of weed nuisance in the river should this be required. These trials should be carried out under stable river flow conditions, and should include higher resolution measurements of diquat, enabling rates of diquat dispersion/adsorption after application to be determined.
- Increase riparian plantings to a height equal to the wetted width in more open sections of the river, ideally on both banks, ensuring plantings create a dense screen.
- Ensure regular calibration of the stage/discharge relationship at Radcliffe Road as this can be strongly affected by weed growth.

[6] For example, McCombs et al. (1999).

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Appendix A Weed biovolume maps

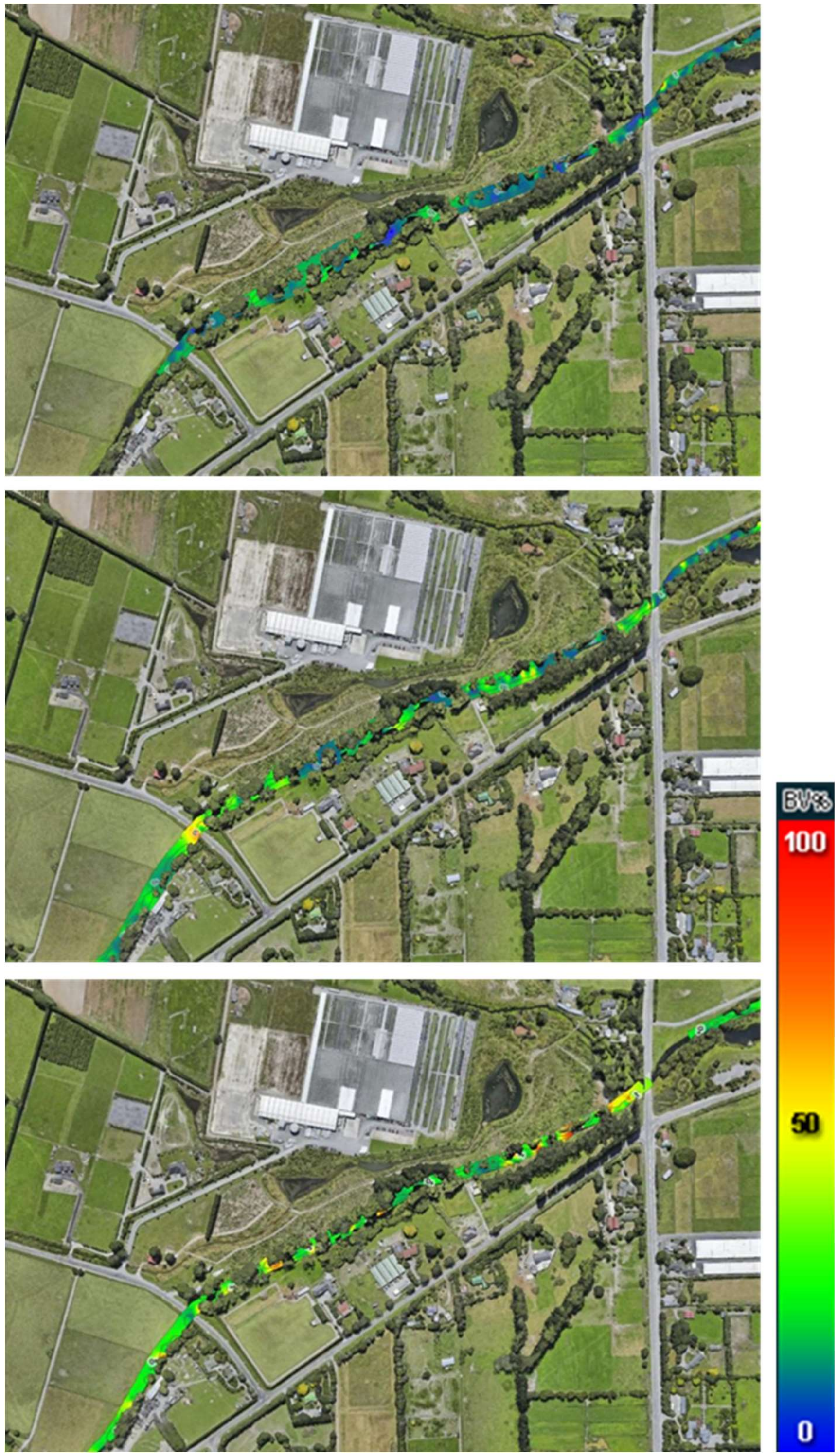


Figure A-1: Weed biovolume maps for the first segment of the River Styx downstream of Redwood Springs. River segment from Radcliffe Rd to Janet Stewart Reserve in November 2020 (top), February 2021 (middle) and April 2021 (bottom). Biovolume key shown to the right.

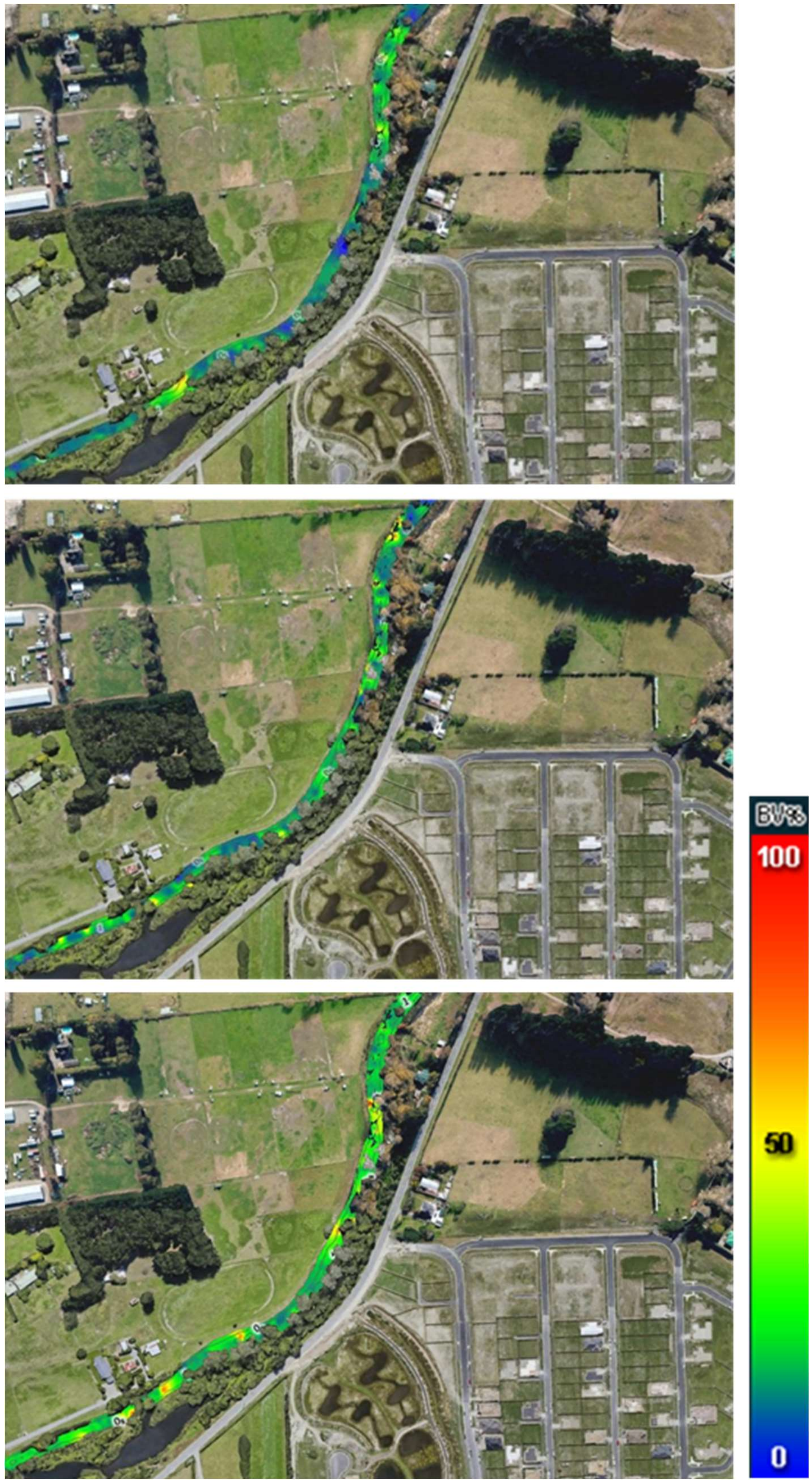


Figure A-2: Weed biovolume maps for the second segment of the River Styx downstream of Redwood Springs. River segment from Janet Stewart Reserve to Dunlop Road (right) in November 2020 (top), February 2021 (middle) and April 2021 (bottom). Biovolume key shown to the right.

Figure A-3: Weed biovolume maps for the third segment of the River Styx downstream of Redwood Springs. River segment from Dunlop Road to Turners Loop in November 2020 (left), February 2021 (centre) and April 2021 (right). Biovolume key shown to the right.

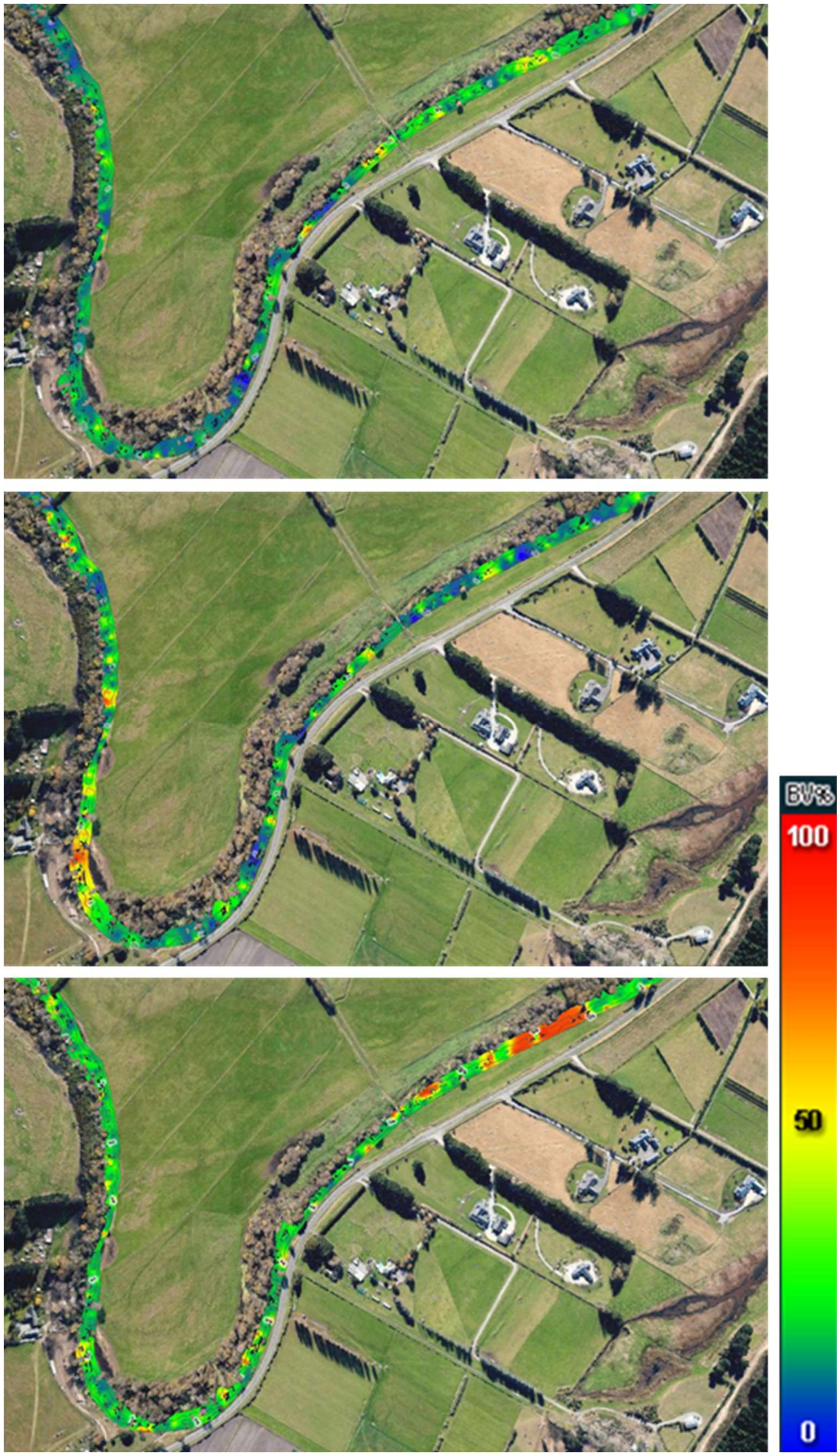


Figure A-4: Weed biovolume maps for the fourth segment of the River Styx downstream of Redwood Springs. River segment from Turners Loop to Lower Styx Road in November 2020 (top), February 2021 (centre) and April 2021 (bottom). Biovolume key shown to the right.

Figure A-5: Weed biovolume maps for the sixth segment of the River Styx downstream of Redwood Springs. River segment from Spencerville Road Bridge to Reserve in November 2020 (left), February 2021 (centre) and April 2021 (right). Biovolume key shown to the right.

Figure A-6: Weed biovolume maps for the seventh segment of the River Styx downstream of Redwood Springs. River segment from Reserve to Kainga Road in November 2020 (left), February 2021 (centre) and April 2021 (right). Biovolume key shown to the right.

Appendix B Harvesting and water level data

Table B-1: Harvesting operation start dates, duration and water level reductions achieved. From data supplied by Christchurch City Council.

Year	Early season start date	Late season start date/s	Early season harvesting duration (days)	Late season harvesting duration (days)	Radcliffe early season water level reduction (m)	Radcliffe late season water level reduction (m)	Lower Styx early season water level reduction (m)	Lower Styx late season water level reduction (m)
2000-01	7/06/00	24/04/21	23	52	0.23	0.20	0.3	0.60
2001-02	30/11/01	23/04/02	59	55	0.20	0.25	0.2	0.30
2002-03	6/12/02	9/06/03	46	50	0.25	0.15	0.3	0.45
2003-04	30/11/03	7/04/04	57	54	0.10	0.40	0.15	0.30
2004-05	31/12/04	31/05/05	39	60	0.13	0.28	0.15	0.55
2005-06	14/02/06	29/07/06	42	52	-	0.42	0	0.20
2006-07	19/12/06	15/05/07	32	46	0.20	0.65	0.02	0.22
2007-08	30/06/07	31/03/08	56	62	-	0.25	-	0.05
2008-09	19/11/08	28/03/09	52	54	0.25	0.45	0	-0.10
2009-10	26/11/09	30/04/10	53	39	0.15	0.23	0.15	0.25
2010-11	11/11/10	27/04/11	55	69	-0.08	0.50	0	0.25
2011-12	5/11/11	21/03/12	56	79	-0.08	0.18	-0.1	0.15
2012-13	20/09/12	23/01/13, 15/05/13	48	92	0.25	0.35	0.23	0.68
2013-14	7/11/13	21/05/14	132	48	-0.10	0.28	-0.2	0.00
2014-15	27/11/14	14/04/15	57	49	-0.04	0.73	-0.06	0.31
2015-16	22/12/15	4/05/16	51	56	-0.10	0.28	0.2	0.60
2016-17	20/09/16	18/01/17, 12/05/17	35	117	-0.10	0.51	0.02	0.09
Mean (±SE)			53 (±6)[a]	61 (±5)[a]	0.08 (±0.04)[a]	0.36 (±0.04)[b]	0.08 (±0.04)[a]	0.29 (±0.05)[b]

"-" denotes missing data
alphabetic superscripts (a, b) indicate significant differences between early and late season water level reductions (paired *t*-test, $p<0.05$).

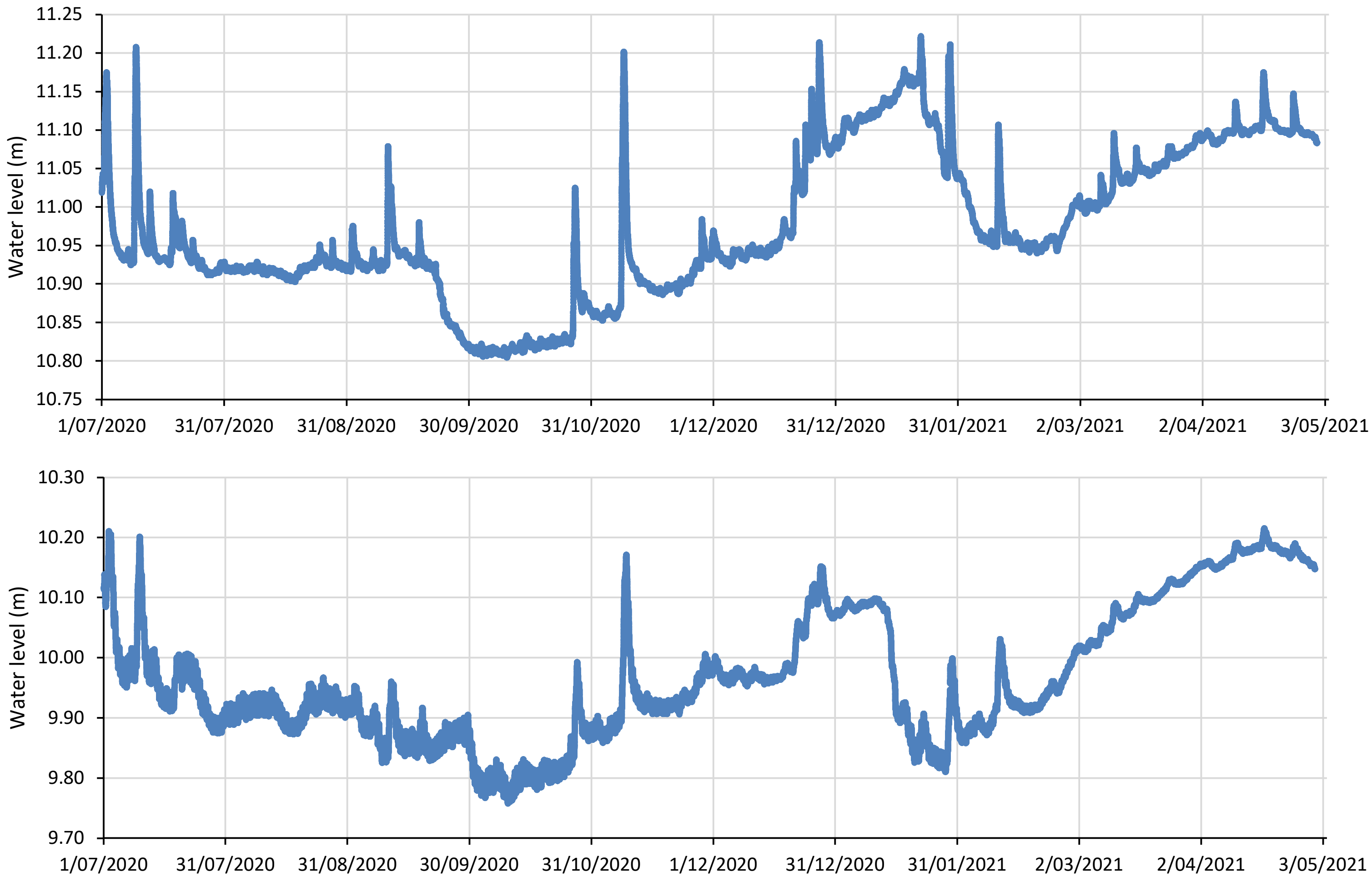


Figure B-1: Water levels at Radcliffe Road and Lower Styx Road from 1 July 2020 to 30 April 2021. Top: Radcliffe Road. Bottom: Lower Styx Road.

Appendix C Diquat trial cover data and photo time-series

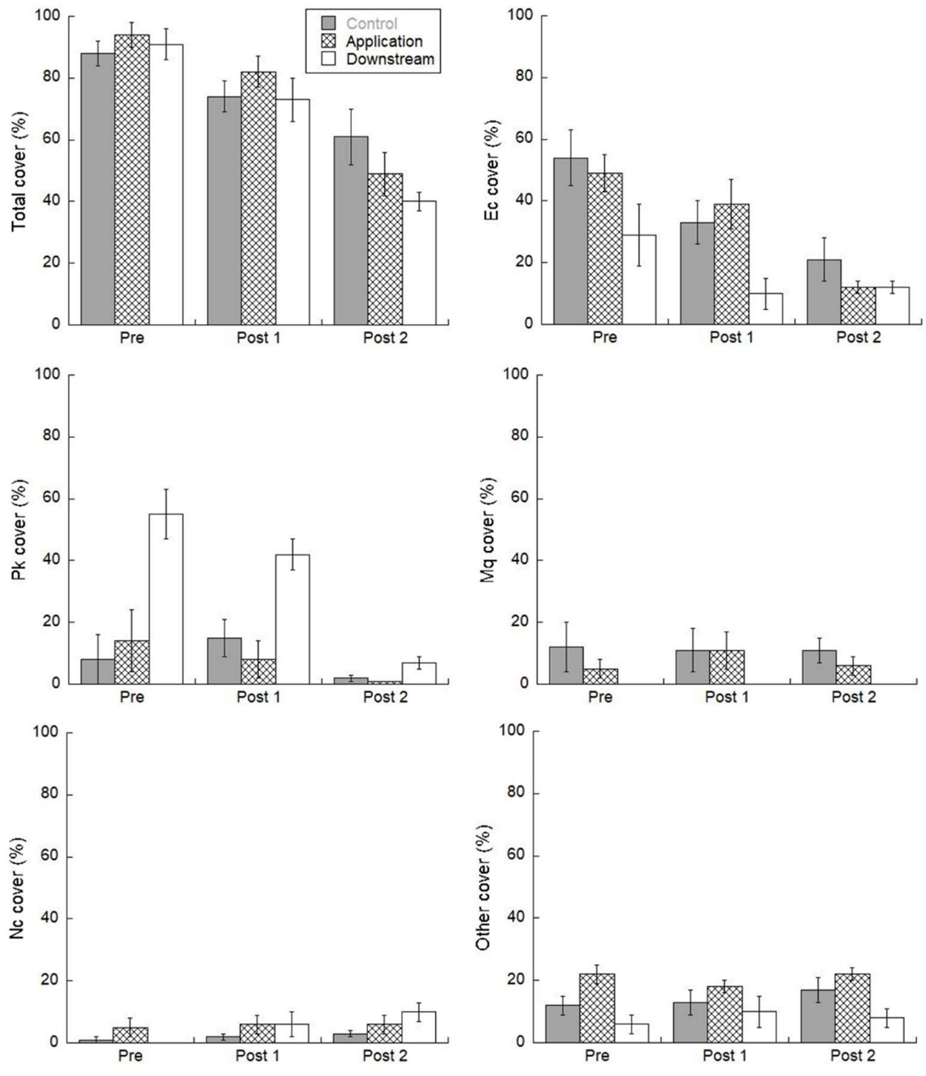
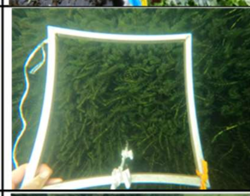
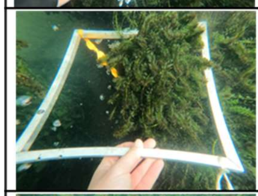
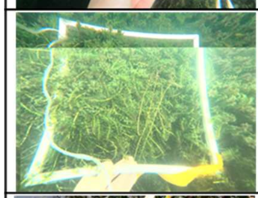
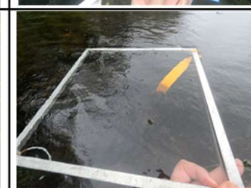
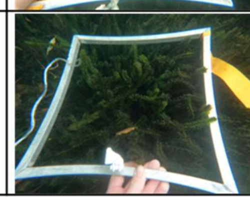
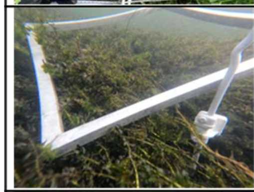
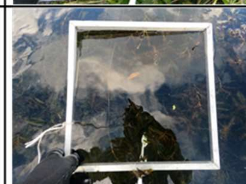
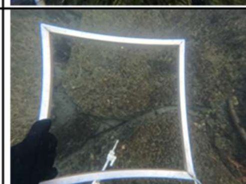
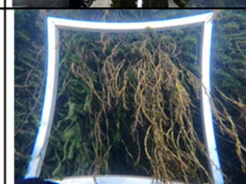
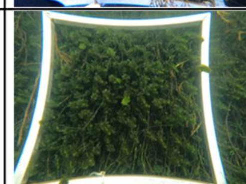
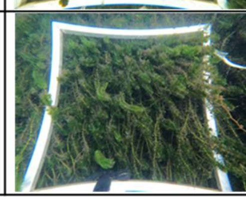


Figure C-1: Cover of aquatic plants, total and by species, before and after diquat application in control, application, and downstream reaches. Values shown are means with bars representing the standard error (n=5). Pre = 21-22 March 2022 survey, Post 1 = 20-21 April 2022 survey, Post 2 = 13-14 June 2022 survey. Ec = *Elodea canadensis*, elodea. Pk = *Potamogeton crispus*, curlyleaf pondweed. Mq = *Myriophyllum propinquum*, common water milfoil. Nc = *Nitella* aff. *cristata*, stonewort. Other = all other aquatic plant species present.

Control reach pre-diquat (21-22 March 2022)				
1	2	3	4	5
				
				
				
				
		No photo		No photo

Control reach post diquat 1 (20-21 April 2022)				
1	2	3	4	5
				
				
				
				
		No photo		

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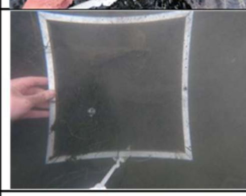
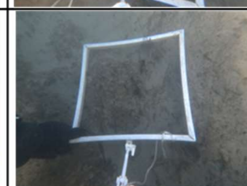
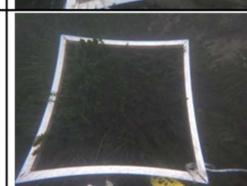
				
	No photo			
				

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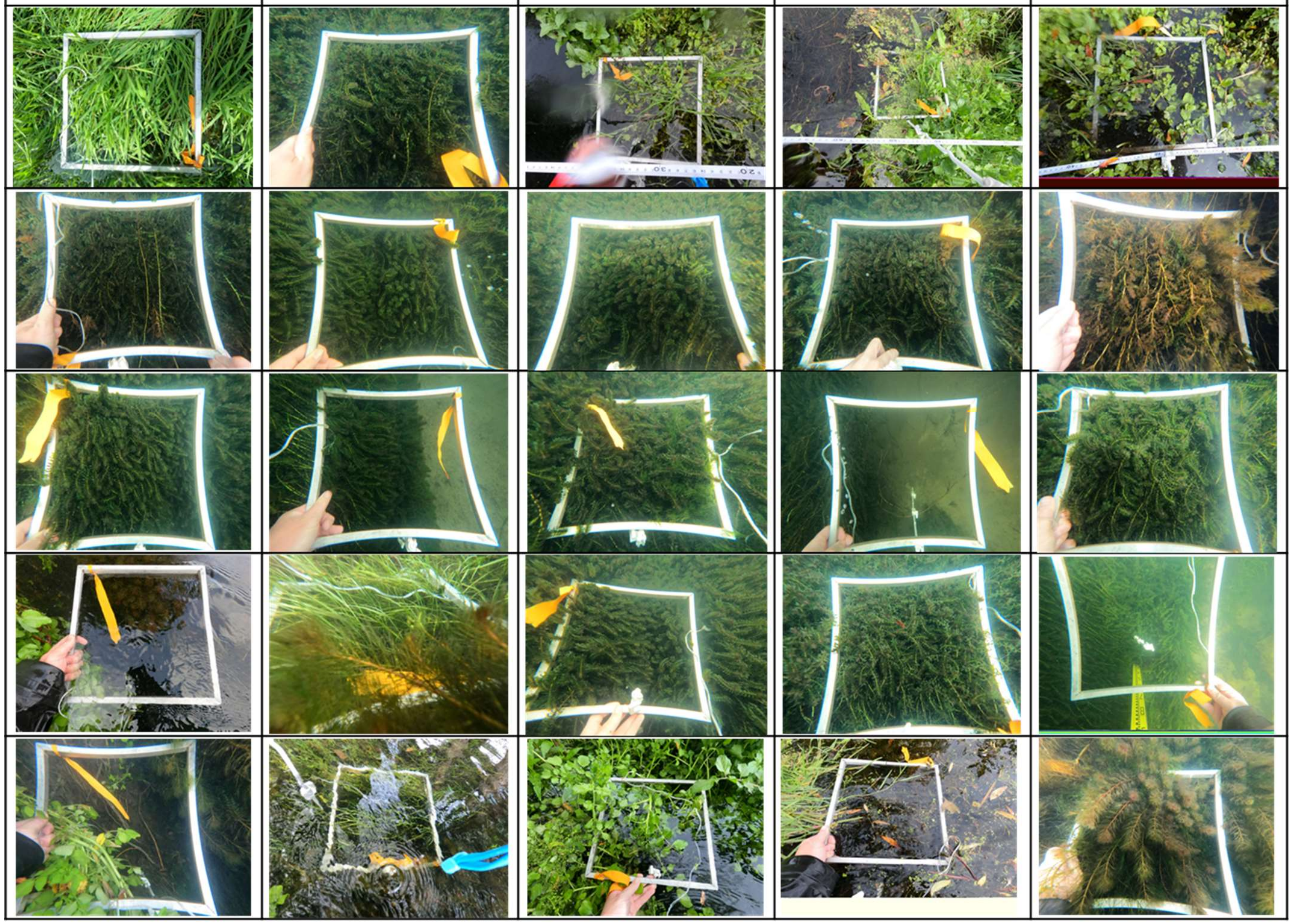

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			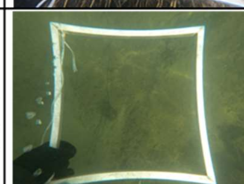	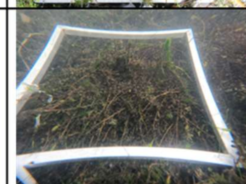
			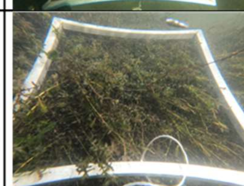	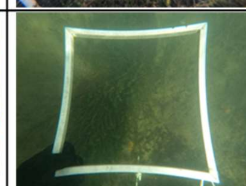
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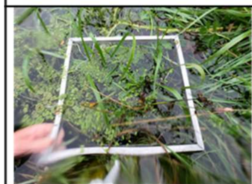	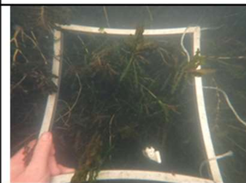	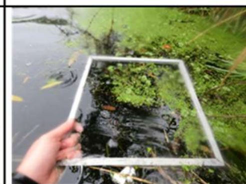		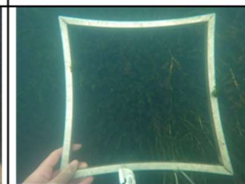
			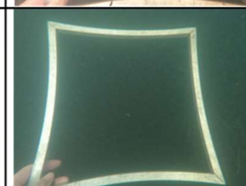	
	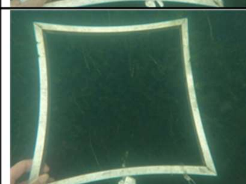	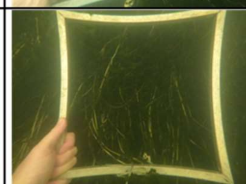	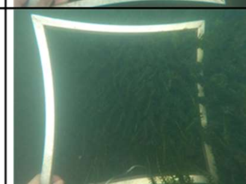	
			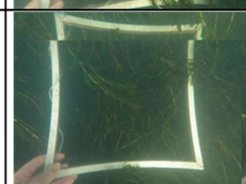	
		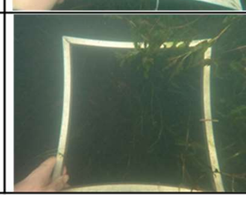		

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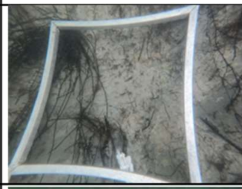
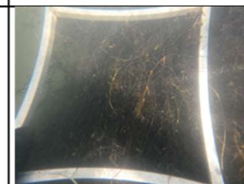
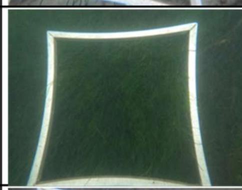
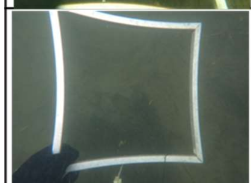
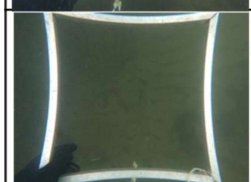
			No photo	
				
				
				
				

Downstream reach post diquat 2 (13-14 June 2022)				
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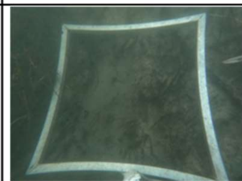
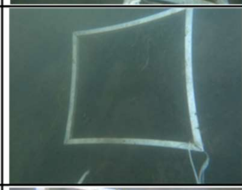
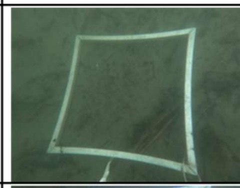
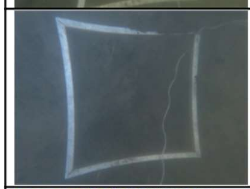
				
		No photo		
				
				
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Figure C-2: Photo time-series from diquat trial survey cross-sections. Three time-series for each reach.

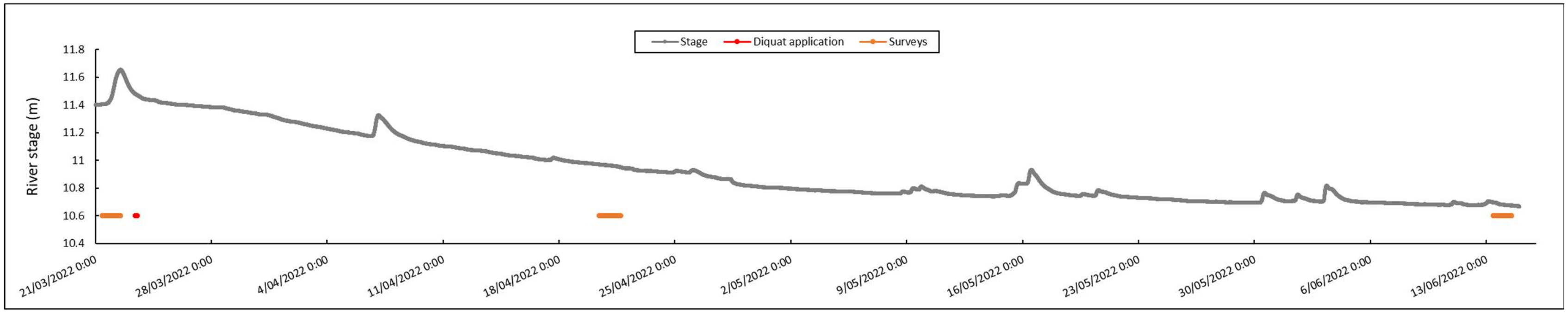

Figure C-3: River stage in the river during the diquat trial. Stage record from Environment Canterbury Radcliffe Road monitoring station. Orange markers indicate when aquatic plant surveys were undertaken.

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Appendix D Shade assessments

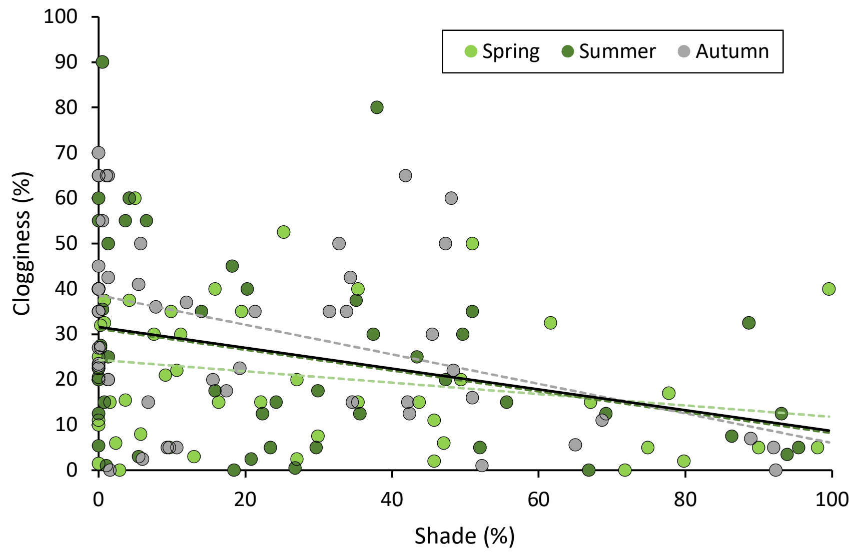

Figure D-1: Relationship between macrophyte clogginess and shade measured by densiometer based on cross-section point data . Black line is for all data across the three survey dates. Seasonal regression lines are the lighter dashed lines shown. Regression relationships are statistically significant at p>0.05 (summer, autumn, overall) and p<0.10 (spring).

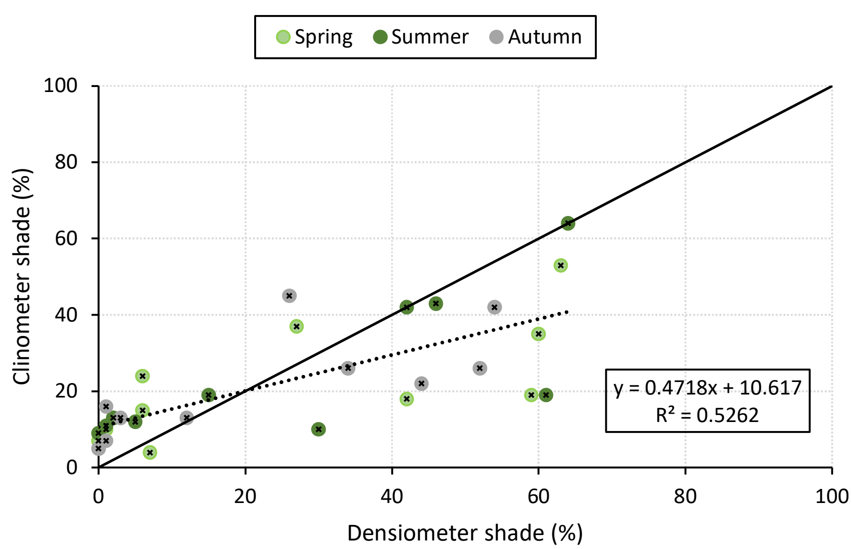

Figure D-2: Shade measured by densiometer versus clinometer for river cross-sections. Clinometer measurements indicated slightly more shade at low levels of shade than the densiometer, and somewhat less shade at higher levels.

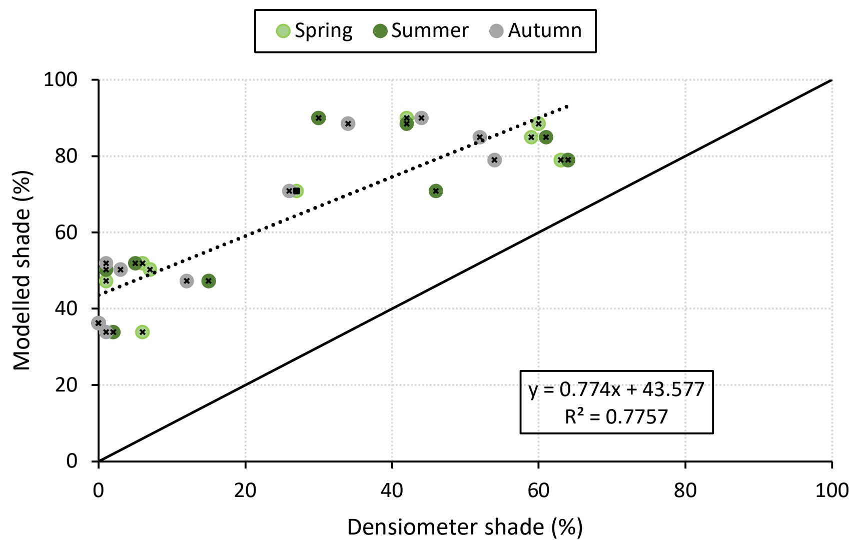

Figure D-3: Shade measured by densiometer versus modelled shade for river cross-sections. Modelling predicted more shade than was measured by densiometer by 15 to 60 % (average= 38 %).

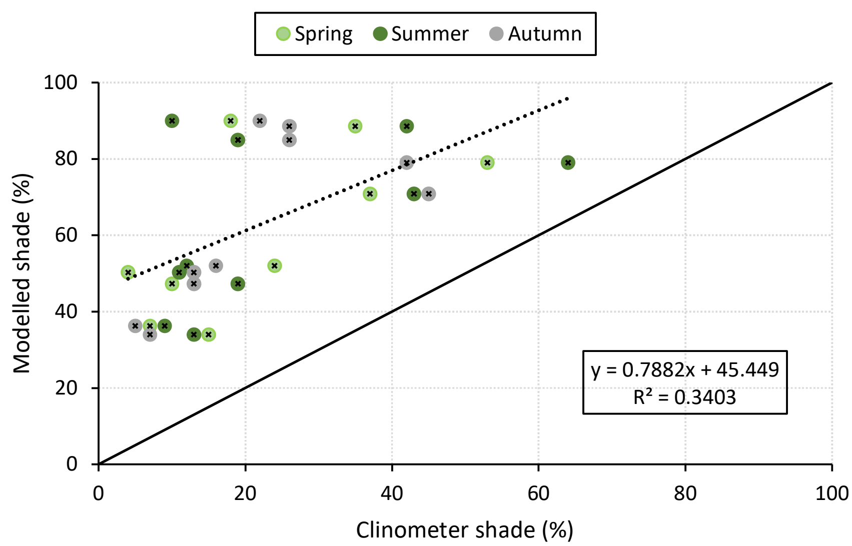

Figure D-4: Shade measured by clinometer versus modelled shade for river cross-sections. Modelling predicted more shade than was measured by densiometer by 15 to 80 % (average= 41 %).

Appendix E Nutrient results and guidelines

Table E-1: Water sampling results summary. Values are means (standard error) of spring, summer and autumn sampling visits.

Site number	Site name	NO_3-N (µg/L)	NH_4-N (µg/L)	DRP (µg/L)	Turbidity (FNU)	Conductivity (µS/cm)
1	Redwood Springs	250 (±14)	8 (±3)	12 (±1)	1.3 (±0.3)	146 (±35)
2	Willowview Drive	277 (±15)	7 (±2)	12 (±1)	1.3 (±0.3)	110 (±3)
3	Radcliffe Rd	397 (±21)	10 (±7)	16 (±3)	1.3 (±0.3)	113 (±5)
4	Janet Stewart Reserve	499 (±32)	13 (±4)	18 (±3)	1.6 (±0.4)	142 (±20)
5	Dunlop Homestead	503 (±55)	11 (±6)	21 (±6)	1.1 (±0.1)	118 (±6)
6	Turners Loop	378 (±27)	17 (±12)	24 (±10)	2.7 (±0.9)	119 (±4)
7	Lower Styx Rd	343 (±76)	13 (±7)	22 (±6)	1.5 (±0.5)	119 (±4)
8	Spencerville Rd	277 (±57)	16 (±10)	28 (±10)	1.9 (±0.8)	118 (±3)
9	Boat Ramp Reserve	251 (±67)	26 (±10)	28 (±15)	5.5 (±3.2)	165 (±42)
10	Kainga/Harbour Rd	244 (±65)	23 (±13)	34 (±11)	2.6 (±0.9)	135 (±13)

Table E-2: Nitrogen and phosphorus contents of River Styx surficial sediments. Nitrogen and phosphorus contents of River Styx surficial sediments.

Site number	Site name	TN (%)	TP (%)
1	Redwood Springs	0.22 (±0.16)	0.04 (±0.00)
2	Willowview Drive	0.27 (±0.06)	0.07 (±0.02)
3	Radcliffe Rd	0.39 (±0.06)	0.17 (±0.02)
4	Janet Stewart Reserve	0.35 (±0.27)	0.08 (±0.05)
5	Dunlop Homestead	0.42 (±0.21)	0.08 (±0.05)
6	Turners Loop	0.40 (±0.15)	0.11 (±0.04)
7	Lower Styx Rd	0.90 (±0.13)	0.12 (±0.03)
8	Spencerville Rd	0.57 (±0.20)	0.16 (±0.09)
9	Boat Ramp Reserve	0.14 (±0.04)	0.04 (±0.00)
10	Kainga/Harbour Rd	0.30 (±0.09)	0.13 (±0.09)

Table E-3: Annual mean dissolved inorganic and total sediment nutrient categories for an instream macrophyte BBN designed to predict the probability of nuisance growth. Reproduced from Matheson et al. (2012). A category can be selected based on only water column or sediment concentrations if only one is available. If one of the nutrients (e.g., DIN) is in the high category and the other is in the adequate category (e.g., DRP) then select the category corresponding to the lowest concentration. If either nutrient is in the limiting category then this category should be selected as growth is likely to be constrained by this nutrient.

Category	Water column nutrients (mg m^{-3} or µg/L)	Sediment nutrients (%DW)	Probability of nuisance growth
High	DIN>1000 and/or DRP >100	TN>2 and/or TP >0.2	0.9 (90%)
Adequate	DIN 100-1000 and/or DRP 10-100	TN 0.1-2 and/or TP 0.01-0.2	0.7 (70%)
Limiting	DIN <100 or DRP <10	TN <0.1 or TP <0.01	0.3 (30%)

Appendix F Water quality and climate trends

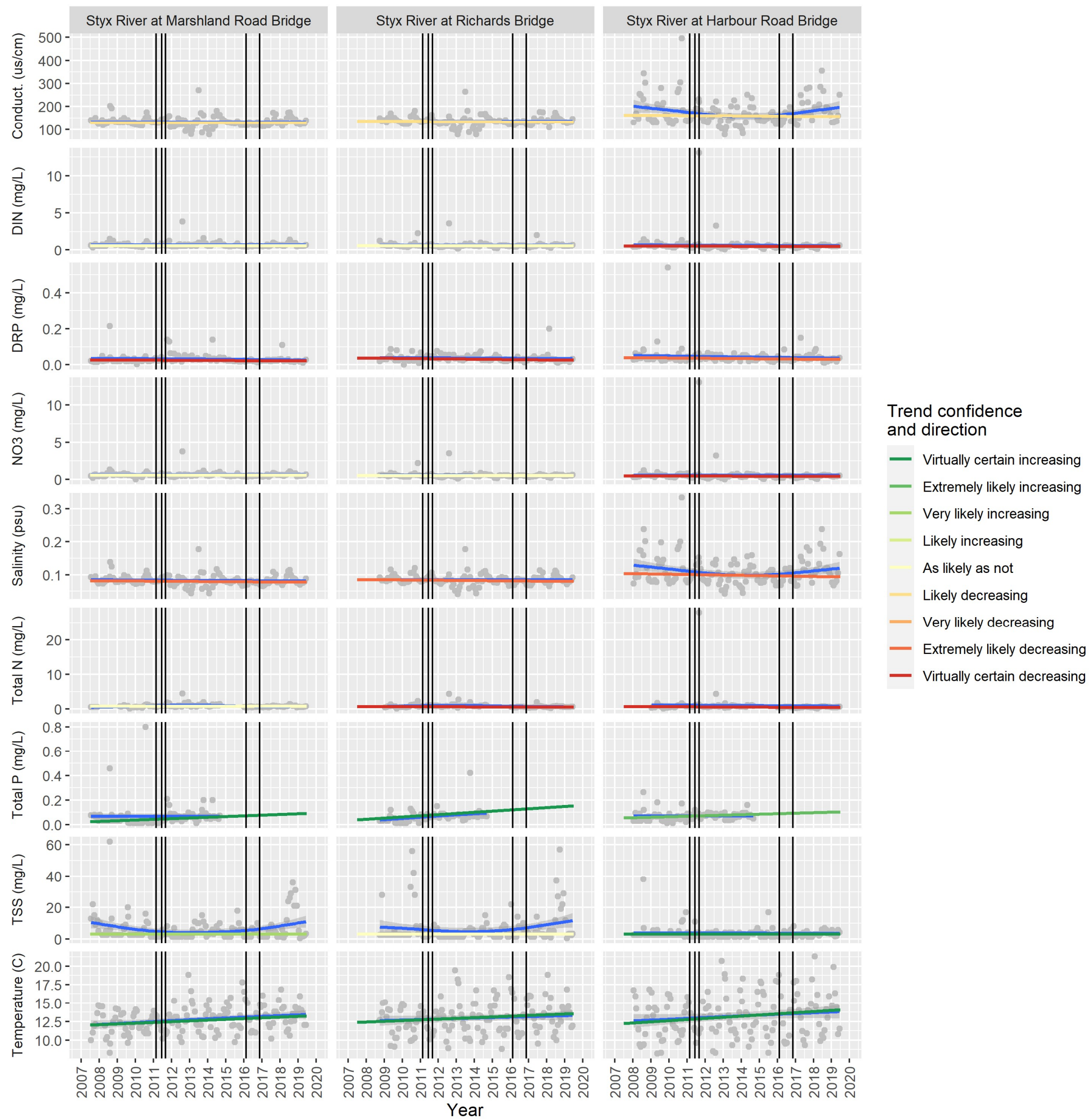

Figure F-1: Water quality trends for Styx River monitoring sites without flow adjustment. Sites ordered left to right from upper to middle to lower river. Vertical lines show significant earthquake occurrences from 2011 (4 September 2010 earthquake not shown). Blue line shows a flexible LOESS curve fit to the data for comparison with the monotonic trend detection line.

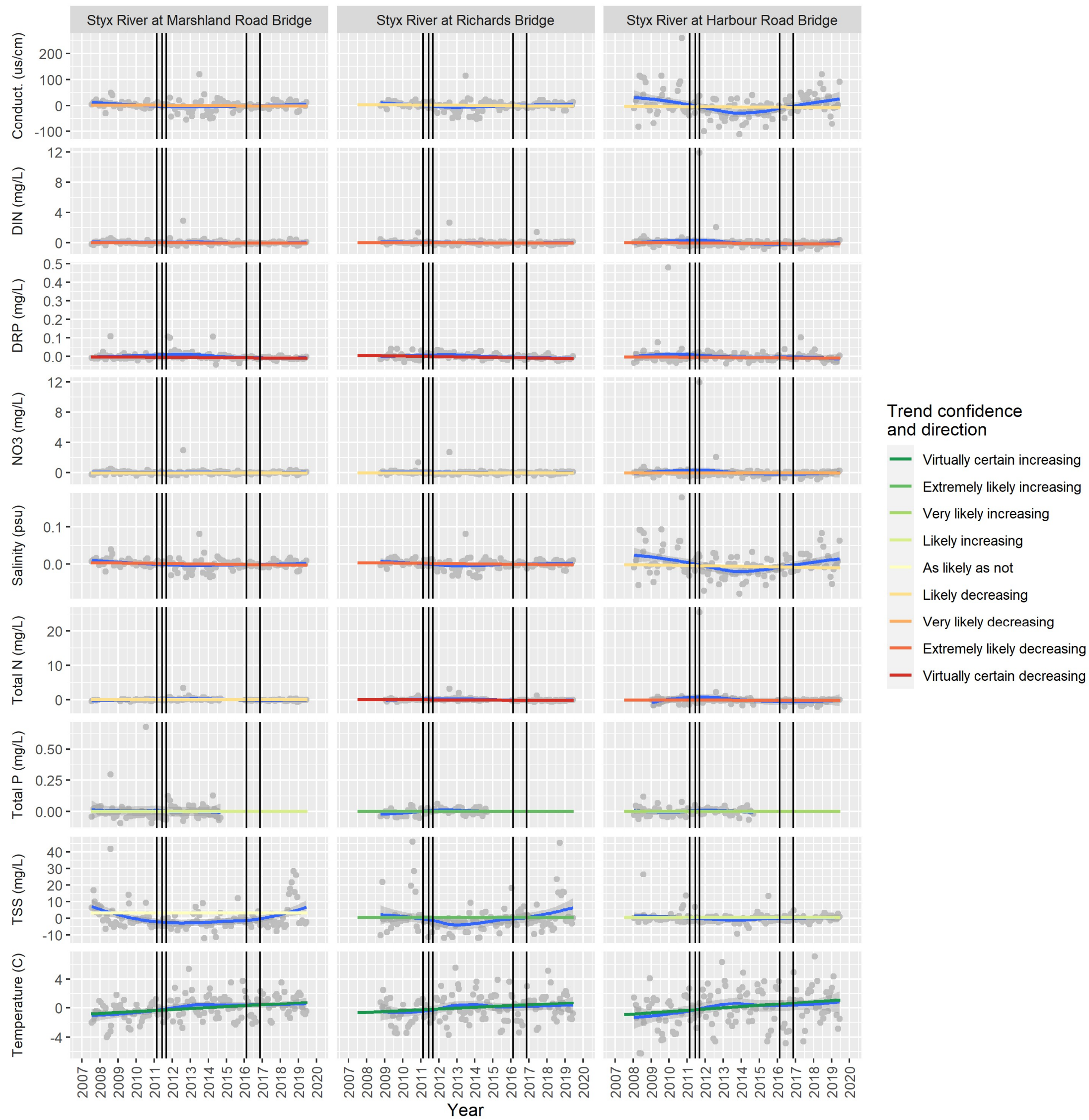

Figure F-2: Water quality trends for river main-stem monitoring sites with flow adjustment. Sites ordered left to right from upper to middle to lower river. Vertical lines show significant earthquake occurrences from 2011 (4 September 2010 earthquake not shown). Blue line shows a flexible local polynomial regression (LOESS) curve fit to the data for comparison with the monotonic trend detection line.

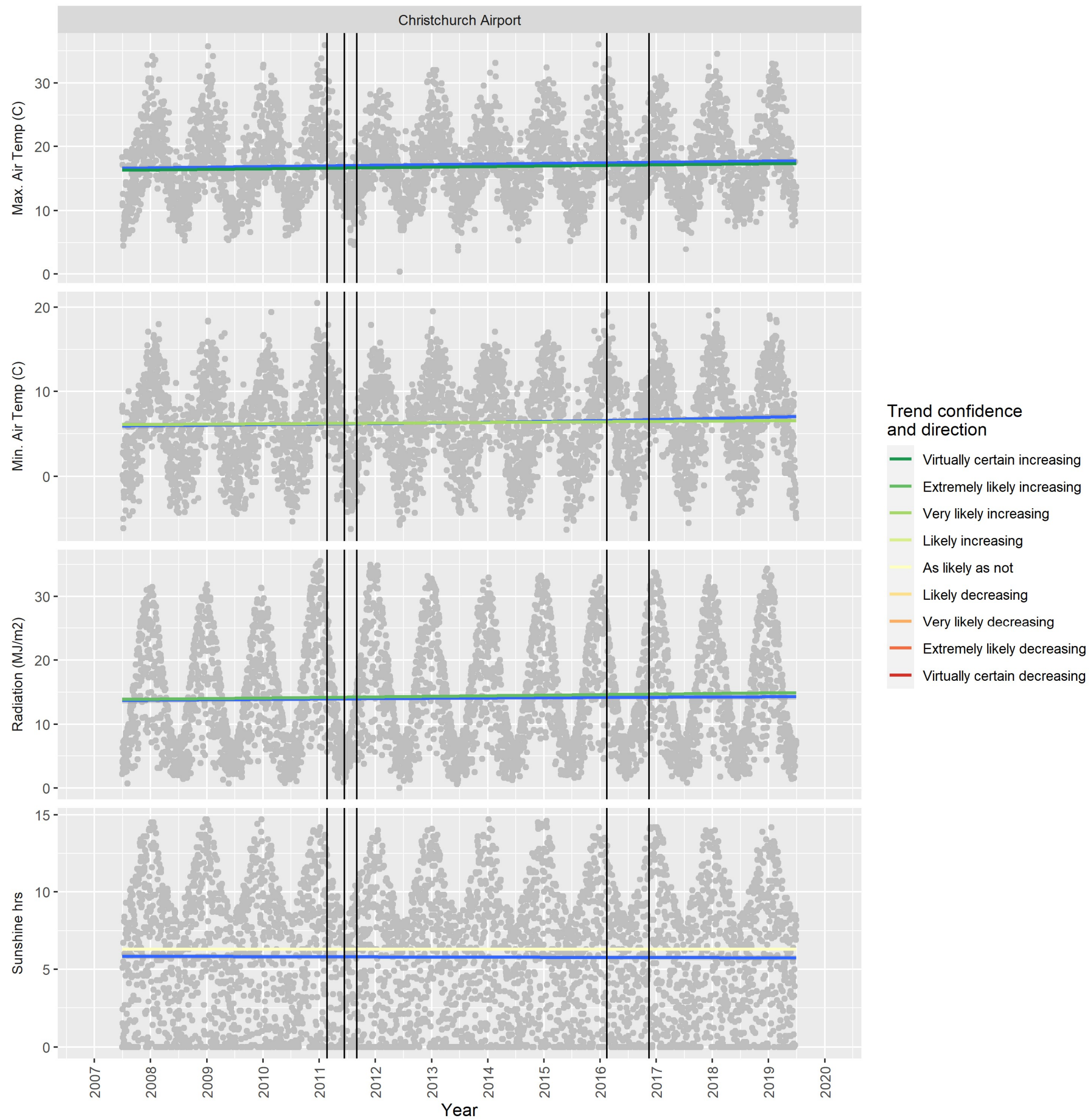

Figure F-3: Trend analysis of climate data from nearby Christchurch Airport from mid-2007 to mid-2019. From top to bottom: Daily maximum air temperature, daily minimum air temperature, daily radiation, and daily sunshine hours. Vertical lines show significant earthquake occurrences from 2011 (4 September 2010 earthquake not shown). Blue line shows a flexible Generalized Additive Model (GAM) curve fit to the data for comparison with the monotonic trend detection line.

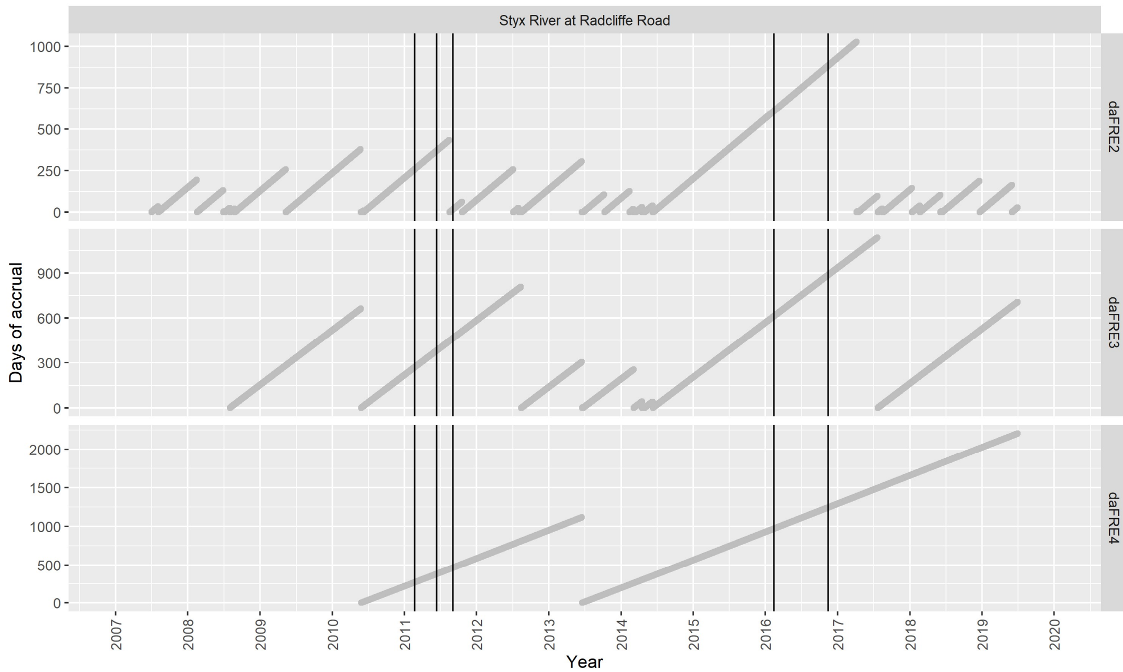

Figure F-4: Days of accrual between flushing flow events. Flow events two (top), three (middle) and four (bottom) times the median flow. Calculated from Radcliffe Rd daily flow records from mid-2007 to mid-2019. After a flow event days of accrual reset to zero. Vertical lines show significant earthquake occurrences.