Interactions between aquatic vegetation, hydraulics and fine sediment

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Running head:
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Abstract
Aquatic vegetation, hydraulics and sediment transport have complex interactions that are not yet well understood. These interactions are important for sediment conveyance, sediment sequestration, phasing of sediment delivery from runoff events, and management of ecosystem health in lowland streams. To address this knowledge gap detailed field measurements of sediment transport through natural flexible aquatic vegetation are required to supplement and validate laboratory results. This paper contributes a field study of suspended sediment transport through aquatic vegetation and includes mechanical removal of aquatic vegetation with a weed cutting boat. It also provides methods to quantify vegetation cover through remote sensing with Unmanned Aerial Vehicles (UAVs) and estimate biomass from ground truth sampling. Suspended sediment concentrations were highly dependent on aquatic vegetation abundance, and the distance upstream that had been cleared of aquatic vegetation. When the study reach was fully vegetated (i.e. cover >80%), the maximum recorded SSC was 14.6 g/m^3 (during a fresh with discharge of 2.47 m^3/s), during weed cutting operations SSC was 76.8 g/m^3 at 0.84 m^3/s (weedcutting boat 0.5-1 km upstream from study reach), however following weed cutting operations (4.6 km cleared upstream), SSC was 139.0 g/m^3 at a discharge of 1.52 m^3/s. The data indicates that fine sediment was being sequestered by aquatic vegetation and likely remobilised after vegetation removal. Investigation of suspended sediment spatial dynamics illustrated changes in particle size distribution due to preferential settling of coarse particles within aquatic vegetation. Hydraulic resistance in the study reach (parameterized by Manning's n) dropped by over 70% following vegetation cutting. Prior to cutting hydraulic resistance was discharge dependent, while post cutting hydraulic resistance was approximately invariant of discharge. Aerial surveying captured interesting changes in aquatic vegetation cover, where some very dense regions of aquatic vegetation were naturally removed leaving behind unvegetated riverbed and fine sediment.

1 Introduction
Fine sediments and aquatic vegetation are interconnected components of lowland river systems. Aquatic vegetation alters river hydraulics through vegetation flow resistance (Aberle & Järvelä, 2015; Bal et al., 2011), which decreases mean velocities, increases water depth, reduces near bed

turbulence, creates preferential flow paths and enhances sedimentation rates (Asaeda, Rajapakse, & Kanoh, 2010; Franklin, Dunbar, & Whitehead, 2008; Wharton et al., 2006;). Sedimentation around aquatic vegetation occurs due to increased residence time for water compared to the free stream (Schulz, Kozerski, Pluntke, & Rinke, 2003) and decreased resuspension of sediment, due to reduced stresses (Vargas−Luna, Crosato, & Uijttewaal, 2015). This can result in preferential deposition of fine sediments under plant canopies (Bennett, Pirim, & Barkdoll, 2002; Jones, Collins, Naden, & Sear, 2012; Wharton et al., 2006). Fine sediments, particularly of organic composition, often have high concentrations of nutrients (Barko, Gunnison, & Carpenter, 1991; Schulz et al., 2003), with implications for plant growth rates, since many plants uptake nutrients through their roots (Chambers, Prepas, Bothwell, & Hamilton, 1989). The growth of aquatic vegetation also creates a downstream-displaced feedback loop, where a decrease in suspended sediment concentration (SSC) due to deposition leads to improved visual water clarity, promoting light penetration and further enhancing growth rates in downstream reaches (Madsen, Chambers, James, Koch, & Westlake, 2001). At catchment scales, aquatic vegetation serves as a partial sediment filter, sequestering a portion of the sediment load delivered from erosion sites during runoff events and temporarily reducing delivery to receiving waters such as lakes and estuaries.

Aquatic vegetation at low-medium biomass is generally considered beneficial for aquatic environments, however at high biomass it can act as an ecosystem engineer, causing bank erosion, changing channel morphology, reducing flow conveyance and increasing flood risk (Biggs et al., 2018; Butcher, 1933; Dunderdale & Morris, 1996; Gurnell, Van Oosterhout, De Vlieger, & Goodson 2006). Abundant aquatic vegetation may also cause excessive fine sediment deposition which can lead to additional environmental problems by making river gravels less permeable to water and thus reducing the flux of dissolved oxygen into the bed. This has negative consequences for salmonid eggs (Argent & Flebbe, 1998; Greig, Sear, & Carling, 2005) and macroinvertebrates (Jowett, 2003; Wood & Armitage, 1999; Wood, Toone, Greenwood, & Armitage, 2005). Therefore, removal of some aquatic vegetation may be necessary for both practical engineering reasons and river ecosystem health (Dawson, 1989; Bal & Meire, 2009). Potential aquatic vegetation removal techniques include flushing flows (Tena, Vericat, Gonzalo, & Batalla, 2017), chemical control (Murphy & Barrett, 1990), biological control with herbivorous fish (Pipalova, 2006; Van der Zweerde, 1990), and mechanical removal (Wade, 1990).

Mechanical removal of aquatic vegetation is common practice globally, with equipment ranging from simple hand tools to mechanised diggers and weed cutting boats (Gettys, Haller, & Bellaud, 2014; Hudson & Harding, 2004; Madsen, 2000; Wade, 1990). Mechanical removal of vegetation is expensive (Dawson, 1989; Madsen & Wersal, 2017; Rockwell, 2003), and can be ecologically detrimental (Garner, Bass, & Collett, 1996; Greer, 2014), with the impact being dependent on removal strategy and frequency (Baattrup−Pedersen & Riis, 2004; Greer, Closs, Crow, & Hicks, 2012). Vegetation removal with mechanised diggers can significantly increase SSC, with the effects lasting for months and SSC reaching levels that are harmful to fish (Greer, Hicks, Crow, & Closs, 2016; Kemp, Sear, Collins, Naden, & Jones, 2011). Weed cutting boats are becoming an increasingly popular alternative to clearing with mechanised diggers. However, it is not well-established what effect these have on SSC, particle size distributions and aquatic ecology. Further information is also needed on how suspended sediment propagates through aquatic vegetation patch mosaics and how far it may travel before settling. Resolving these knowledge gaps will assist with the design of vegetation cutting strategies that try to balance ecosystem health with practical engineering considerations, such as maintaining river conveyance and limiting sedimentation.

This study aims to investigate the effect of aquatic vegetation on river hydraulics and on spatio-temporal dynamics of fine sediments. The effect of aquatic vegetation management on suspended sediment concentrations and particle size distributions were also investigated by comparing the measurements before and after vegetation removal in the Halswell River located in Canterbury, New Zealand.

2 Study reach, data collection and methods

2.1 Study reach

Field measurements were performed in the Halswell River from January to May of 2018 during the southern hemisphere summer and autumn. The river drains into Te Waihora (Lake Ellesmere). The mean annual flood at the monitoring site downstream of the study reach (Ryans Bridge site) is 6 $m^3 s^{-1}$, with mean flow of 1.3 $m^3 s^{-1}$ with 4 event per year exceeding three times the median flow (FRE 3), the annual suspended sediment load is approximately 2270 tonnes (Hicks, Semadeni-Davies, Haddadchi, Shankar, & Plew, 2019). The Halswell River is predominantly spring fed and generally has a stable flow regime outside of heavy rain events. The area of catchment upstream of the study reach is 204 km^2. The 432 m long study reach (Figure 1) has a low gradient of 0.000167 and low sinuosity. It was densely vegetated with *Potamogeton crispus* and *Elodea canadensis*. The nutrient and fine sediment load in the Halswell river originates from agricultural land, hill slopes and urban areas.

(Figure 1)

2.2 Data collection strategy

Data collection was designed to investigate the effect of aquatic vegetation on river hydraulics and spatio-temporal fine sediment dynamics. Temporal dynamics were addressed by a 3 month monitoring campaign that covered a range of flow conditions, along with growth and removal of aquatic vegetation with a weedcutting boat. Measurements consisted of UAV flights for aquatic vegetation cover, suspended sediment sampling and ADCP deployments for site hydraulic conditions (Figure 2). These measurements were undertaken to address questions such as: How does aquatic vegetation abundance influence suspended sediment concentrations and particle size distributions? How does aquatic vegetation change site hydraulic conditions? Are fine sediments sequestered or remobilised during freshes in the presence of aquatic vegetation? What effect does a weed cutting boat have on suspended sediment concentrations and particle size distributions? Are sequestered sediments remobilised after aquatic vegetation are removed? Spatial dynamics were addressed by a 1 day high resolution sediment resuspension study (Figure 2), where samples were taken at 3 locations downstream from the resuspension location (Figure 1d). These measurements were undertaken to address questions such as: How do suspended sediment concentrations and particle size distributions change as a function of distance traveled downstream through aquatic vegetation biomass? How far may different size fractions be expected to travel before being sequestered?

(Figure 2)

2.3 Sediment measurements

Temporal SS dynamics

Suspended sediment samples to investigate temporal dynamics were collected at a bridge at the upstream end of the study reach (Figure 1d) using a pole-deployed depth integrating isokinetic sampler (US-DH48) at the centre of the channel where there was no aquatic vegetation present to interfere with sampling. Since the Halswell River was approximately a rectangular channel with width ranging from ~4-6 m, samples from one vertical were representative of the whole cross section. Samples at the upstream bridge were taken in triplicate: one for analysis of Suspended Sediment Concentration (SSC), Volatile Suspended Sediment Concentration (VSSC), and Inorganic Suspended Sediment Concentration (ISSC); one for particle size distribution analysis; and the other frozen as a backup. Samples were collected at 19 times between the 1st of February and 30th April 2018.

Spatial SS dynamics

To investigate suspended sediment spatial dynamics (8th February 2018), a suspended sediment source was created near the upstream end of the study reach (Figure 1d). This was achieved by a team member in the river removing aquatic vegetation and kicking up fine sediment. During this experiment, suspended sediment samples were obtained at three downstream locations spaced approximately equidistantly every 150 m along the study reach (Figure 1d). Depth-integrated samples were collected from an inflatable boat in the centre of the river using a DH48 sampler. Sampling from a boat enabled depth integrated samples to be obtained from the centre of the river without any interference from a human operator stirring up sediment. 36 samples were taken for SSC, VSSC and ISSC analysis, with another 9 samples taken for particle size analysis.

Processing SS samples

The ASTM standard test method (D3977-97) for measuring sediment concentration for water samples was used in this study. Collected water-sediment mixture samples were first passed through 63 μm sieve. Sediments coarser than 63 μm were oven-dried and their weight measured to determine suspended sand concentration. Sediment concentration on residue of sediments finer than 63 μm were determined by filtering the samples using 1.5 μm pore size filters, and then oven-drying at 104 ˚C temperature. Sediment masses were then measured to determine suspended mud concentration. Combining sand (> 63 μm) and mud (< 63 μm) concentration gives total SSC. The detection limits in this method for measuring sand and mud concentrations were 1 g/m^3 and 0.5 g/m^3, respectively. The volatile suspended sediment concentration for sand and mud concentrations were obtained from the loss on ignition of the mass of measured sand and mud concentration by furnacing the samples at a temperature of 400

˚C. Particle size analysis was performed using an EyeTech Particles Size and Shape Analyzer which uses a combination of laser obscuration and image analysis techniques.

2.4 Discharge

A discharge record at the study site was proxied off the record provided by Environment Canterbury (ECan) at their Ryans Bridge gauging station (N -43.662, E 172.542). The gauging station was ~5 km downstream from the study site, with the catchment area increasing by 14% between the two sites (Figure 1c). Discharge measurements were performed with a Teledyne Stream Pro ADCP during the period from the 1st of February to the 12th of April 2018 at the downstream boundary of the study site (Figure 1d). ADCP measurements were performed in a cross section that was cleared of aquatic vegetation. Each discharge measurement was the average of at least 4 repeated ADCP cross sections, with uncertainties reported as two times the standard deviation of measured discharge. ADCP discharge measurements at the study site displayed a consistent relationship with those recorded downstream by ECan (Figure 3a). Differences in discharge due to the spatial separation of the two sites (i.e. physical processes such as groundwater inputs and water abstraction) were accounted for using the equation $Q_{Ryans Bridge} = 1.15 \times Q_{Study\ Site} - 0.106 \left(m^3 s^{-1}\right)$ (Figure 3a), which was used to generate a study site discharge record (Figure 3b). The close correspondence between the increase in downstream discharge of 1.15 (i.e. +15%) and increase in catchment area (+14%) likely reflects consistent rainfall over the relatively small spatial scale of the Halswell catchment. The relationship between measured discharge at the study site and rated discharge at the ECan site did not change substantially with vegetation abundance, presumably due to ECan keeping aquatic vegetation cleared around their gauging site thereby avoiding vegetation-induced complications to the stage-discharge rating there. Stage was also recorded continuously at the study site and so could have been used to provide a discharge record directly, however this was impacted by vegetation growth, vegetation dynamic reconfiguration (as a function of flow conditions) and passage of the weed cutting boat. Thus, the ECan-based discharge record was preferred as we consider that it provided a more accurate flow record. The small-scale, high-frequency ripple in the discharge data (Figure 2) was investigated with Fourier analysis, and was found to be highly periodic. Distinct energy peaks corresponded to signals with 24 hr period and harmonics with 12 hr, and 48 hr period. This is likely due to water abstraction for irrigation since the magnitude of the ripple is higher during the driest months of summer.

(Figure 3)

2.5 Hydraulics

The study reach was a single thread rectangular lowland channel (Figure 1d) with little variation in width and cross sectional mean depth throughout the study reach. Bulk hydraulic conditions at the study site were evaluated from the ADCP measurements at the downstream end of the study reach in a cross section cleared of aquatic vegetation. Cross sectional mean velocity was calculated by dividing total discharge Q by cross-sectional area A. Aquatic vegetation flow resistance was parameterised through Manning's resistance coefficient *n*, where $n = \frac{1}{U} R_h^{2/3} S_f^{1/2}$

(Graf & Altinakar, 1998) with hydraulic radius $R_h = A/P = A/(b+2h)$ where b is river width and h is river depth. The friction slope S_f was taken to be equivalent to water surface slope, since channel cross-section shape and h were approximately uniform throughout the study reach. The width to depth ratio b/h was less than 10 for all measurements before vegetation cutting. The total boundary shear stress due to friction was parameterised by $\tau_0 = \frac{\rho g S_f dV}{dA} = \frac{\rho g S_f bhdL}{(b+2h)dL} = \rho g S_f R_h$ where ρ is water density and g is gravitational acceleration (Henderson, 1966). For vegetated channels it should be remembered that 'total boundary shear stress' is purely a metric for flow resistance and has little relevance to sediment transport processes since most of the flow resistance arises from drag on aquatic vegetation which is distributed throughout the water column and transfers drag forces to the bed through their stems. The rate of energy dissipation was parameterised by stream power per unit bed area $SP_A = \rho g Q S_f / b = \tau_0 U$ (Bagnold, 1966) and stream power per unit volume $SP_V = \rho g Q S/(bh) = \rho g U S$ (Biggs et al., 2018). Stream power per unit volume is a useful metric for vegetated flows where drag forces are distributed throughout the volume of river water and can be used to evaluate aquatic vegetation hydraulic habitat preferences.

Water surface slope was determined by the principle of hydrostatic levelling (Gordon, Finlayson, & McMahon, 2004) using a manometer that was 99.3 m long. The manometer was fitted with an electric pump ('Whale Submersible 12V pump', model GP1352) to flush the hose between measurements and remove any air bubbles that may have come out of solution. Slope measurements were centred at the midpoint of the study reach (Figure 1d), with 4 independent measurements taken and averaged (average slope of the study reach was 0.000169). Repeated slope measurements (with manometer flushing between measurements) were generally very consistent, with discrepancies of only 0-1 mm deviation in elevation head between each repeated measurement. The use of a manometer is preferable to a piezometer, which can include a small amount velocity head as it is hard to find locations without any velocity for the upstream inlet (Biggs et al., 2018). Piezometer inlets also suffer from clogging, are more susceptible to bubbles in the line and are harder to flush. The only downside of manometers is that the distance from the reference water level to the river surface must be read at both the upstream and downstream ends, doubling this source of measurement uncertainty.

2.6 Aquatic vegetation

Aquatic vegetation abundance in the study reach was measured in terms of planform area, cover and fresh biomass.

Aerial cover surveys

Aerial surveys of macrophyte cover in the study reach were performed with a DJI Phantom 4 Pro UAV ('drone'). Ground Control Points (GCPs) were distributed throughout the study reach and surveyed with a Trimble Real-Time Kinematic (RTK) GPS system. Aerial images were processed in Agisoft Photoscan Pro (i.e. Agisoft Metashape) to generate georeferenced orthomosaics. At least 10 GCPs were used for each aerial survey. Georeferenced orthomosaics were then imported into ArcMap, where delineation of wetted channel boundaries and segmentation of the wetted channel into classes 'aquatic vegetation' and 'unvegetated riverbed' were performed manually. This provided data on aquatic vegetation planform area (m^2) throughout the study reach at 6 survey times covering the late summer vegetation growth and mechanical weed cutting (Figure 2). For further details of aquatic vegetation aerial surveying methods see Biggs et al., (2018) and Biggs, (2020).

Shapefiles of polygon boundaries around classes and a polyline for the river centreline were drawn manually in ArcMap. Data synthesis, analysis and visualisation was performed using MATLAB 2018b (MathWorks, Inc). Polygons of each class were converted to raster maps with a grid resolution of 5 cm (0.0025 m^2 cells). The manually defined river centreline was smoothed and resampled to 0.2 m intervals (in natural coordinates). Unit vectors orthogonal to the centreline were used to define the sides of bins that spanned the river width [Appendix 1]. These bins were then used to sum the area of each class of interest within the bin (from the raster maps). Bins were then summed with downstream distance. The summation of the raster data was compared to summation of the original polygons from each class to check accuracy and discretisation errors. For the class 'aquatic vegetation' the error in the study reach was -0.148 m^2 (with percentage error of -0.00768%), for the class 'unvegetated riverbed' the error in the study reach was +0.209 m^2 (with percentage error of +0.0610%). These errors are very small and indicate that the selected 5 cm grid resolution was sufficiently fine. For comparison, a grid resolution of 0.5 m produced errors of +7.064 m^2 (+0.366%) for 'aquatic vegetation' and -2.841 m^2 (-0.827%) for 'unvegetated riverbed'. Aquatic vegetation cover is the area of 'aquatic vegetation' divided by the area of wetted riverbed.

Biomass

For the investigation of spatial SS dynamics (8th of February 2018) aquatic vegetation biomass was quantified throughout the study reach. This was achieved by establishing a conversion factor from planform area to biomass (known as biomass areal density with units of kg/m^2), which could then be multiplied by planform area throughout the study reach to estimate the spatial distribution of aquatic vegetation biomass (Biggs, 2020). To achieve this 27.0 m^2 of riverbed (upstream from where ongoing aerial monitoring commenced Figure 1d) was manually cleared of aquatic vegetation, which was taken to the laboratory where it was centrifuged to remove surface water, then weighed. Vegetation centrifuging was performed in a 9.5 kg top loading washing machine (Simpson Eziset 950) on the repeatable spin cycle (Biggs, 2020), similar to the approach of Edwards and Owens (1960).

The area of riverbed that aquatic vegetation was removed from was determined from aerial imagery taken before and after vegetation removal, with removed vegetation patches demarcated manually with polygons in the georeferenced orthomosaics. The upstream area that was cleared of aquatic vegetation contained a mixture of *Potamogeton crispus* and *Elodea canadensis* at densities and proportions that were representative of biomass throughout the study reach. The biomass metric used was kg of fresh weight (after centrifuging to remove surface water) which was suitable for the large volumes of biomass collected from the study reach, compared to kg of dry weight, or g of ash free dry weight, which are suitable for smaller biomass samples (Biggs, 2020).

3 Results

3.1 Aquatic vegetation cover and biomass

Aquatic vegetation cover increased from 78.8% on the 1st of February to 92.4% on the 8th of March. A slight reduction in cover to 88.7% was measured on the 28th of March, before aquatic vegetation was removed by the Environment Canterbury weed cutting boat on the 5th of April (Figure 4).

For the investigation of spatial suspended sediment dynamics on the 8th of February, 27 m^2 of upstream riverbed was manually cleared of aquatic vegetation. The fresh weight of this aquatic vegetation was 70.3 kg and provided a biomass aerial density of 2.60 kg/m^2. This value was used to estimate biomass distributions throughout the study reach during the spatial suspended sampling campaign (Figure 5). The average biomass per unit channel length was 12.43 kg/m.

(Figure 4)

(Figure 5)

3.2 Hydraulics

Hydraulic resistance generally increased from the 1st of February to the 28th of March in association with vegetation growth, then decreased dramatically following the vegetation cut (Figure 6). Cross sectional mean velocity in the study reach decreased with increasing vegetation cover, going from 0.120 m/s on the 1st of February at 0.649 m^3/s to 0.102 m/s on the 28th of March at the higher discharge of 0.712 m^3/s. Following the vegetation cut [6th of April at 0.844 m^3/s], cross sectional mean velocity jumped to 0.234 m/s, Manning's *n* decreased from 0.107 to 0.0315 (-70.5%), total boundary shear stress decreased by 44.4%, and stream power per unit volume increased by 128.9%.

Manning's *n* was discharge dependent when aquatic vegetation was present but was less sensitive to discharge after the vegetation was removed (Figure 7). For example, Manning's *n* decreased by 24.2% during an early February event with maximum flow of 1.40 m^3/s but changed little during the larger fresh from the 10th - 14th of April (Q_{max}= 2.17 m^3/s) following vegetation removal.

(Figure 6)

(Figure 7)

3.3 Suspended sediment temporal dynamics

Suspended sediment temporal dynamics, including the 'sediment rating curve' relationship between SSC and discharge, were highly dependent on aquatic vegetation abundance upstream of the study reach and activities of the weed cutting boat (Figure 8, Figure 9). When the study reach was vegetated SSC remained below 15 g/m^3, even during runoff events. During the 21st of February 2018 event, measured SSC was only 14.6 g/m^3 at a discharge of 2.47 m^3/s compared to 139.0 g/m^3 during a lower flow of 1.52 m^3/s on the 30th of April, once the entire upstream channel had been cleared of aquatic vegetation.

(Figure 8)

When the weed cutting boat was actively cutting upstream (Figure 2 and green circle in Figure 9) SSC jumped to 76.8 g/m^3 at a discharge of 0.84 m^3/s, which is 5.26× the maximum sampled SSC before any cutting. The effect of weed cutting on SSC was also dependent on the length of channel upstream that was cleared. After all weed cutting operations were complete (4.6 km cleared upstream), SSC was 139.0 g/m^3 at a discharge of 1.52 m^3/s (pink circle in Figure 9), which is 9.52× the maximum sampled vegetated SSC. When the upstream reach was only partially cleared, suspended sediment concentrations of only 50.2 g/m^3 were measured near the peak of a larger event at a discharge of 2.02 m^3/s (Figure 8, Figure 9). While likely also influenced by the temporal variation and phasing of the SSC in the runoff delivered from the catchment headwaters during these events, this difference illustrates the impact of upstream aquatic vegetation on trapping and retaining fine sediment.

(Figure 9)

Suspended sediment transport through the study reach was dominated by inorganic material during the runoff events and following vegetation removal (Figure 10). The peak suspended sediment load before vegetation clearing was 36.1 g/s during ex-tropical cyclone Gita (21/02/2018) at a discharge of 2.47 m^3/s. After upstream weed cutting had been completed a suspended sediment load of 210.8 g/s was measured at a discharge of only 1.52 m^3/s.

(Figure 10)

From the 1st of February to the 8th of March the particle size of suspended sediment in the study reach generally decreased (Figure 11) as aquatic vegetation cover increased (Figure 4). From the 8th to 28th of March suspended sediment size increased slightly as aquatic vegetation cover decreased. The size of suspended sediment particles increased dramatically during aquatic vegetation cutting on the 5th of April. The size distribution of suspended sediment also remained coarser after vegetation cutting had been completed. On the 30th of April (11 days after vegetation cutting had been completed) the size distribution was D10=16.1 µm, D50=49.2 µm, D90= 94.6 µm at a discharge of 1.52 m^3/s, which was more coarse than that observed on the 21st of February (D10=14.6 µm, D50=40.1 µm, D90=63.8 µm) at the higher discharge of 2.47 m^3/s when the river was vegetated.

(Figure 11)

3.4 Suspended sediment spatial dynamics during sediment supply experiment

Total suspended sediment and inorganic suspended sediment loads decreased with downstream distance and cumulative biomass of aquatic vegetation that the suspended sediment passed through (Figure 12). Sediment load decreased from approximately 6.31 g/s to 3.07 g/s after travelling 304.2 m downstream and passing ~3700 kg of aquatic vegetation biomass (1418 m^2 of aquatic vegetation planform area) that was distributed over 1679 m^2 of wetted riverbed. For the sediment load, flow conditions, and aquatic vegetation in the study reach, this provides approximate trapping rates of $8.76*10^{-4}$ (g/s)/kg of aquatic vegetation biomass, or $2.28*10^{-3}$ (g/s)/m^2 of aquatic vegetation planform area, or $1.93*10^{-3}$ (g/s)/m^2 of wetted riverbed area.

The composition of the SS load also changed with downstream distance and cumulative biomass. The inorganic component of the load decreased from 79.18% to 70.39% and the volatile component of the load increased from 20.82% to 29.61%. The particle size of the total suspended sediment (dominated by inorganic material) also decreased with downstream distance and cumulative aquatic vegetation biomass (Figure 13, Figure 14). Thus coarse, inorganic particles were preferentially deposited, with D90 decreasing from 58.56 μm to 29.27 μm, D50 decreasing from 36.08 μm to 15.64 μm, and D10 decreasing from 13.39 μm to 5.96 μm, after travelling 304.2 m downstream and passing ~3700 kg of aquatic vegetation biomass. The uncertainties in the SS load likely reflect variability in SS supply, compared to a fixed rate injection system, however, particle size measurements were largely unaffected since they are not dependent on SSC.

(Figure 12)

(Figure 13)

(Figure 14)

4 Discussion

4.1 Cover and biomass

The decrease in measured cover from 92.4% to 88.6% from the 8th to 28th of March (Figure 4) cannot be accounted for by vegetation hydraulic removal during floods, as no high discharge events occurred during that period (Figure 4). However, there were regions of the study reach which were formerly very densely vegetated, that had their vegetation completely removed during this period (Figure 15). The reason why this occurred is unknown as there were no weed cutting activities during this period and riverbanks were fenced to limit livestock access to the river. It is possible that extensive sedimentation within the dense macrophyte stands at the river margins may have exceeded the burial tolerance of these species (Jones et al., 2012), or decreased the rooting strength of 'dense vegetation clusters' as vegetation becomes primarily attached to fine sediment (Biggs *et al* 2018). However, *Elodea canadensis* and *Potamogeton crispus* both produce adventitious roots (Sculthorpe, 1967), which should make them more adaptable to burial by fine sediment. It is also possible that dissolved oxygen spatial variations and microbiological processes may have contributed to the degradation and removal of dense regions of aquatic vegetation. For example, the dense regions of *P. crispus* in Figure 15a may have suffered from insufficient fluxes of light, water and dissolved oxygen (particularly overnight). This could have caused localised hypoxia and enhanced anaerobic microbial activity, causing tissue damage to stems. A small perturbation in flow direction (as aquatic vegetation continues to grow in other parts of the river) may have resulted in a cascade of vegetation removal, through a combination of erosion of fine sediment from shallowly rooted stems, with shearing and breakage of degraded stems that are attached to deeply buried rhizomes (Bilby, 1977). This hypothesis may explain a natural self-limiting feedback mechanism for vegetation abundance in the Halswell River, and other rivers around the world, where similar patterns of vegetation growth and removal are observed. Further studies of oxygen content and microbial activity within vegetation (and sediment) in these densely vegetated regions would be informative to investigate these processes. Likewise, continuous monitoring of selected study sites with fixed video cameras to visualise vegetation removal processes would be illuminating.

(Figure 15)

4.2 Hydraulics

The Halswell River had similar slope and hydraulic characteristics to other silty lowland streams (Dunderdale & Morris, 1996; Helmiö, 2004; Nikora et al., 2008). Hydraulic resistance in the study reach (parameterised by Manning's *n*) varied between 0.060 and 0.107 in vegetated conditions (Figure 7c), which corresponds to the range of 0.050 to 0.120 predicted by Chow, (1959) for an 'Excavated or Dredged Channel' with the 'Channel not maintained, weeds and brush uncut' and 'Dense weeds, high as flow depth'. Manning's *n* values in this range are also common for natural vegetated streams and rivers (Biggs et al., 2018; Nikora et al., 2008), although higher gradient drains and small streams may have *n* values exceeding 0.500 when they are choked with aquatic vegetation (Champion & Tanner, 2000; Nikora et al., 2008). The dependence of hydraulic resistance in the study reach on discharge (Figure 6b, 2-3rd February) was likely due to reductions in cross sectional blockage as depth increased, and reductions in vegetation cross sectional area at higher velocities, due to streamlining and vegetation reconfiguration closer to the bed (Aberle & Järvelä, 2015; Gosselin & de Langre, 2011). The spatial structure of aquatic vegetation in the Halswell River was quite different from rivers with higher gradient and cross sectional mean velocity, where macrophytes such as *Ranunculus sp.* form a patch mosaic distribution (Biggs et al., 2018; 2019). In the Halswell River aquatic vegetation formed dense stands (predominantly along the river margins) with relatively clear channels meandering through the vegetation. This geometry is similar to a downscaled floodplain with forests of foliated flexible emergent vegetation on either side of the river and lateral exchange of higher momentum (and sediment laden) fluid into the vegetation (Box, Västilä, & Järvelä, 2019; Västilä & Järvelä, 2018). Similar aquatic vegetation structure was also reported by Champion and Tanner, (2000), with new channels forming through the dense aquatic vegetation towards the end of summer. In the Halswell River *P. crispus* biomass dominated throughout the growing season until the vegetation cut in April, which contrasts with the competition dynamics observed by Champion and Tanner, (2000) with *P. crispus* biomass peaking early in the growing season (December), then seeming to be outcompeted by *Egeria densa* as the growing season progressed. Their observations may be due to feedback loops between vegetation growth, hydraulic resistance, depth, cross sectional mean velocity, sedimentation, and species specific habitat preferences (Butcher, 1933; French & Chambers, 1996). For example, Champion and Tanner, (2000) found *P. crispus* most commonly growing in areas with velocities greater than 0.1 m/s, with these sites more prevalent at the start of the growing season, before significant reductions in cross sectional mean velocity and increases in *E. densa* biomass. Since cross sectional mean velocities in our study reach were maintained above 0.1 m/s throughout the growing season (and *E. densa* was not present) it is likely that hydraulic conditions favored *P. crispus* over *E. canadensis*.

The aerial surveying techniques used in the Halswell River were suitable for quantifying 2D distributions of aquatic vegetation cover (Biggs et al., 2018) and how these change over the growing season. However, these methods are limited in how they can resolve vegetation 3D spatial structure, which is essential for predicting hydraulic resistance in vegetated channels (Nikora et al., 2008; Savio, 2017). Techniques for converting from 2D planform area (cover) to biomass or site average 3D structure have been developed by Biggs (2020), however they are specific to a single aerial surveying campaign (due to vegetation growth and changes in relative submergence with discharge). As such, these techniques are suited to single high resolution field deployments (e.g. biomass estimation in this study), rather than long term monitoring campaigns. To address this problem, remote sensing techniques that can resolve 3D vegetation spatial structure are needed. For clear unvegetated rivers it is possible to resolve bathymetry from through water imagery corrected for surface refraction (Dietrich, 2017; Woodget, Carbonneau, Visser, & Maddock, 2015). This technique may also be applicable for bathymetry between macrophyte patches, however problems would likely be encountered when trying to reconstruct the 3D spatial structure of aquatic vegetation due to vegetation motion corrupting point matching when using multi-view stereo algorithms. It would also not be possible to assess bathymetry underneath aquatic vegetation using these techniques, or to apply them in highly turbid rivers. To overcome these limitation cross sectional surveys with acoustic surveying techniques (Mizuno et al., 2018; Sabol, Melton, Chamberlain, Doering, & Haunert, 2002; Stocks, Rodgers, Pera, & Gilligan, 2019) may be more suitable.

4.3 Suspended sediment temporal dynamics

Suspended sediment temporal dynamics in the Halswell River were strongly linked to aquatic vegetation abundance (Figure 8, Figure 9). In rivers that are clear of aquatic vegetation, runoff events typically dominate the temporal variability of SSC, however in the Halswell River, SSC loads during runoff events (such as that on the 21st of February) were strongly damped by the presence of dense aquatic vegetation. This indicates that aquatic vegetation upstream from the sampling site was effective at trapping and retaining fine sediment, with SSC after vegetation cutting likely a function of the length and thus bed area of upstream channel that fine sediment was being resuspended and transported from. For example, SSC was 14.6 g/m^3 at a discharge of 2.47 m^3/s when the river was fully vegetated, SSC was 50.2 g/m^3 at 2.02 m^3/s when aquatic vegetation had been cut from 1,100 m upstream of the study site, and SSC was 139.0 g/m^3 at 1.52 m^3/s when 4,600 m upstream had been cut. This relationship illustrates the potential of aquatic vegetation to be used as an engineering tool to enhance deposition and trap fine sediment in targeted sections of maintained channels.

During weed cutting on the 6th of April (when the weed cutting boat was 500-1000 m upstream of study reach) SSC was 76.8 g/m^3 at a discharge of 0.84 m^3/s. This was 20.76× the SSC measured on the 28th of March when the upstream reach was fully vegetated (3.7 g/m^3 at 0.80 m^3/s). Although the weed cutting boat caused a large jump in SSC, it was a much lower impact than observed from mechanical excavation elsewhere (e.g. Greer et al., 2016), indicating that the use of weed cutting boats may be a more environmentally friendly solution (from a peak SSC perspective) when vegetation removal is necessary.

After vegetation cutting had been completed upstream of the study site suspended sediment concentration reached levels in excess of 100 g/m^3 (i.e. 100 mg/l), which may have negative impacts on fish and environmental health (Berry, Rubinstein, Melzian, & Hill, 2003; Caux, Moore, & MacDonald, 1997). Following vegetation cutting the proportion of suspended sediment comprised of organic matter (volatiles) decreased. This is likely representative of the proportion of organic and inorganic fine sediment that is trapped by aquatic vegetation, then liberated following vegetation cutting. For example, when aquatic vegetation was manually removed from the study reach and fine sediment artificially resuspended to investigate spatial suspended sediment dynamics (Figure 12), inorganic SS suspended sediment comprised at least 65% of TSS.

The particle size distribution after all vegetation cutting had been completed (Figure 12) had D90 of 94.61 µm, D50 of 49.24 µm, and D10 of 16.10 µm. This was remarkably similar to the predicted particle size distribution at the resuspension location from the suspended sediment spatial dynamics study (Figure 13). For example, back projection of particle size from the 3 sampling locations to the sediment source with lines of best fit of the form $y = a e^{-bx}$ as a function of channel centreline downstream distance x (since integrated biomass was approximately a linear function of downstream distance), predicted a D90 of 75.22 µm, a D50 of 48.75 µm and a D10 of 18.10 µm. This indicates that SSC after vegetation cutting was largely comprised of fine sediment that was being resuspended from the channel bed, which was previously trapped by aquatic vegetation. Future work in this direction with a different vegetation cutting regime and a more intensive suspended sediment sampling campaign would be illuminating. For example, if the upstream portion of the channel was cut first it would then act as a reliable suspended sediment source supplying the downstream vegetated section. SSC could then be measured by an array of autosamplers spaced at intervals from the unvegetated/vegetated interface to at least 2 km downstream. This would provide valuable data on suspended sediment transport, deposition and resuspension in vegetated rivers. A continuous SSC record derived from turbidity sensor data would also be extremely valuable to look at through event loads. During this study turbidity sensors were deployed, but suffered from severe biofouling (repeated tangling with aquatic vegetation) and did not provide a reliable record, thus only manually collected SSC samples were used. For future work on sediment dynamics weekly maintenance and inspection of autosamplers and turbidity sensors would be recommended.

4.4 Suspended sediment spatial dynamics

Suspended sediment spatial dynamics depend on aquatic vegetation structure and river hydraulics. For example, species such as *P. crispus* and *E. canadensis* are commonly found in low velocity reaches with fine mud and silt (Butcher, 1933), whereas *Ranunculus* species dominate reaches with higher velocities and coarser substrate (Biggs et al., 2018; Cotton, Wharton, Bass, Heppell, & Wotton, 2006). In the Halswell river, the dense stands of *P. crispus* and *E. canadensis* promoted settling of suspended sediment as it was transported downstream through the vegetation. This was quantified by a decrease in suspended sediment load (Figure 12) and particle size distribution (Figure 13, Figure 14), as river water transited downstream past aquatic vegetation biomass. In the Halswell river it is likely that fine sediment was preferentially transported down flow paths between the dense vegetation, then laterally transported and/or diffused into the dense stands of vegetation where accumulation was observed. This is a similar configuration of vegetation, flow and sedimentation to Cotton et al., (2006) and Box et al., (2019), rather than the isolated vegetation patches of Sand-Jensen (1998), Biggs et al., (2019) and Przyborowski, Łoboda, Bialik and Västilä (2019). At the study sites of Cotton et al., (2006) an accumulation of 6-16 cm of fine sediment was reported within dense macrophyte stands. This sediment was dominated by sand, with less than 10% comprising particles less than 63 microns, compared to the Halswell river where silt and clay dominated. This is likely due to the much steeper gradient of their study sites (slope of 0.0071 and 0.0032) and higher velocities of up to 0.8 m/s around macrophyte stands facilitating transport of coarser particles, then settling within macrophyte stands at velocities as low as 0.05 m/s. The flexible submerged vegetation in the Halswell river was effective at promoting settling of suspended sediment (particularly the coarser fractions), with D50 decreasing from 36.08 μm to 15.64 μm, after traveling 304.2 m downstream. This result contrasts with sediment transport in rigid cylinder arrays, where sediment with grain size between 420 and 600 μm are readily mobilized and transported at cross sectional mean velocities of only 2-4x those observed in the Halswell river (Yang & Nepf, 2019). Rigid cylinders provide a useful geometric analogue for vegetation such as mangrove roots, however they do not adequately characterize the flexibility, dynamic reconfiguration, spatial structure, wide range of length scales and viscous drag of flexible vegetation. These problems likely lead to far higher stem scale turbulence, and near bed turbulent kinetic energy than that observed for flexible foliated vegetation (Box et al., 2019), with consequences for sediment transport and deposition.

5 Conclusions

Aquatic vegetation and its management play an important role in regulating fine sediment temporal and spatial dynamics in lowland rivers. As illustrated here by the Halswell River, suspended sediment concentrations remained below 15 g/m^3 when the study reach was fully vegetated (i.e. cover >80%), even during runoff events, then jumped to 76.8 g/m^3 during vegetation cutting and up to 139.0 g/m^3 during a runoff event after the entire upstream channel had been cleared. Reductions in SSC and particle size distributions as water travelled downstream through aquatic vegetation in the Halswell River study reach show the effectiveness of vegetation at trapping fine sediment. The weed cutting boat had a lower impact on SSC than was originally expected, which indicates that it is a more environmentally friendly solution than mechanical excavation (when removal of aquatic vegetation from rivers is necessary). Aerial surveying of aquatic vegetation was an effective way to quantify changes in vegetation cover over time and detected natural removal of large sections of dense vegetation. Conversions from vegetation planform area to biomass through ground truth sampling were successful, but only provided conversion factors valid for a single survey campaign due to vegetation growth and density changes.

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Disclosure statement

There are no known conflicts of interest related to the work.

Data availability

The data that support the findings of this study are available on request from the corresponding author. The data are not publicly available due to privacy or ethical restrictions.

Appendix: Discretisation of raster remote sensing grid to river natural coordinates

Downstream distances in rivers are commonly reported in natural coordinate systems (i.e. following the river flow). It is also convenient to report other parameters of interest in river natural coordinates (i.e. fish counts per unit downstream distance). For sparse data this is not problematic, however for dense data (i.e. remote sensing data), where the entire river surface is covered in data points, care must be taken with how data is discretised and allocated to a river centreline (i.e. natural) coordinate. Once data are allocated to a river centreline coordinate, they can be integrated along the natural coordinate system to see how the parameter changes with downstream distance. For this paper, aquatic vegetation planform area is the raster data of interest. The challenge of this discretisation is that the sinuosity of rivers can lead to the intersection of bins that are orthogonal to the river centreline. Bin intersection (overlap) means that raster data could be counted multiple times, or introduces ambiguity in the allocation of data to a bin. The centreline interval spacing is a very important parameter to select correctly to avoid intersecting bins. This problem can be avoided by ensuring that all points on the river centreline comply with the criteria $\frac{W_b}{2}<r$ and $W_b>b$ where W_b is the width of the rectangular bins (Figure 16), r is the radius of curvature of the centreline and b is river width. The radius of curvature of the centreline is related to the centreline bin spacing dL as $r=\frac{dL}{d\theta}$ where $d\theta$ is the angle of curvature of the river centreline. In practice, a convenient approach is to set a centreline bin spacing that is 'probably suitable', then check for bin intersection by solving the intersection of line equations $y=m_i x+c_i$ and $y=m_{i+1}x+c_{i+1}$ for each pair of unit normal vectors $\hat{n}_i$ and $\hat{n}_{i+1}$, where $\hat{n}_i=\begin{bmatrix} u_i \\ v_i \end{bmatrix}$, $m_i=\frac{v_i}{u_i}$ and $c_i \text{¿} y_i-m_i x_i$. The intersection of the line equations is at $X_{i,i+1}=\frac{c_{i+1}-c_i}{m_i-m_{i+1}}$ and $Y_{i,i+1}=m_i X_{i,i+1}+c_i \text{¿} m_{i+1} X_{i,i+1}+c_{i+1}$. To ensure adequate centreline bin spacing to avoid intersection the criteria $\frac{W_b}{2} < \sqrt{\left(X_{i,i+1} - x_i\right)^2 + \left(Y_{i,i+1} - y_i\right)^2}$ and $\frac{W_b}{2} < \sqrt{\left(X_{i,i+1} - x_{i+1}\right)^2 + \left(Y_{i,i+1} - y_{i+1}\right)^2}$ should be satisfied for every centreline pair from $i=1$ to $i=N-1$. Parallel lines $m_i = m_{i+1}$ and vertical lines $m_i = \infty$ or $m_{i+1} = \infty$ are unlikely to occur in natural rivers, but can easily be handled as special cases. Algorithms may use centreline unit normal vectors originating from the midpoint of centreline line segments, or may use them originating from each centreline node and correspond to the 'mid angle' between the centreline unit normal vectors from each of the adjacent line segments (Figure 16). For our study reach in a narrow river with low sinuosity a centreline bin spacing of 0.2 was suitable. For wide rivers with high sinuosity a larger centreline bin spacing may be required. Centreline distances should be referenced to the original high resolution centreline, rather than the straight-line distances between bin centres, which will change with selection of centreline bin spacing. A visual inspection of bins throughout the study reach (Figure 16) to check for bin intersection is also good practice and can be perform instead of (or in addition to) the numerical criteria above. Other approaches such as allocation of pixels to the nearest centreline location are also possible, however problems can be encountered with this approach for irregular and highly sinuous rivers.

(Figure 16)

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