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Published in:
Journal of Ecohydraulics
Publication date:
2018

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Download date: 25. Sep. 2020
Coupling UAV and hydraulic surveys to study the geometry and spatial distribution of aquatic macrophytes

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Coupling UAV and hydraulic surveys to study the geometric characteristics and spatial distribution of aquatic macrophytes

Aquatic macrophytes are a key component of river systems around the world. Surveys of macrophyte surface cover, cross-sectional blockage and plant/patch sizes provide data for river managers to assess instream habitat, hydraulic resistance, sediment dynamics, and the success of stream restorations. Manual surveying techniques are labour intensive, provide low spatial detail and are predominantly applied at the cross-section scale, resulting in a lack of published data on macrophyte size distributions. In this study, 1,099 *Ranunculus penicillatus* patches were surveyed using a UAV-mounted digital camera. Geometric properties such as patch area, length, aspect ratio and orientation were determined from the aerial orthophotos. These data were coupled with hydraulic measurements collected using an acoustic Doppler current profiler and a piezometer device. Macrophyte abundance corresponded to specific ranges of velocity, Froude number and stream power, indicating clear patterns of hydraulic habitat use (and preferential modification) by *Ranunculus*. At the reach scale, flow redirection around dense vegetation clusters was observed, with implications for localised sedimentation and bank erosion. The reported data can improve the design of laboratory experiments to represent *Ranunculus* characteristics in the field. The aerial surveying techniques can be used to efficiently estimate vegetation abundance, surface area blockage factor and also to visualise flow through patch mosaics, enabling targeted management of aquatic vegetation.

Keywords: Aquatic vegetation; UAV; drone; macrophyte; aerial imagery; orthophoto; image analysis; stream power; hydraulic control; vegetation management; surveys; ADCP; piezometer; ecohydraulics.

1. Introduction

Aquatic macrophytes are primary producers that play crucial roles within lotic ecosystems. They form a habitat matrix for aquatic invertebrates (Shupryt & Stelzer 2009), provide flow refugia for fish (Figueiredo et al. 2015), affect river hydraulics (Butcher 1933), alter sediment dynamics (Gurnell et al. 2006) and act as ecosystem engineers (Gurnell 2014). The physical structure of macrophytes also contributes to the
diversity of river ecosystems (Jeffries 1993). For example, increased structural complexity of macrophytes can increase the biomass (McAbendroth et al. 2005) and diversity (Taniguchi et al. 2003) of aquatic invertebrates. However, extremely high densities of aquatic macrophytes, stimulated by increased nutrient loadings (Elser et al. 2007) or flow alterations (e.g. flow regulation (Rørsløtt & Johansen 1996; Bunn & Arthington 2002)), can lead to river management problems, including enhanced sedimentation and increased flood risk (Gurnell et al. 2006). High macrophyte densities have also contributed to the proliferation of pest insect species, such as blackfly (Ibáñez et al. 2008, 2012a), which pose human health risks (Enk 2006). To mitigate these problems, nuisance macrophytes such as Ranunculus penicillatus may be mechanically cut and removed from river channels (Bal & Meire 2009) or, in regulated rivers, plants can be removed by specially designed flushing flows (Batalla & Vericat 2009; Tena et al. 2017). However, mechanical removal can have negative ecological consequences, notably decreased abundance of invertebrates and fish (Kaenel & Uehlinger 1999; Garner et al. 1996).

Balancing management considerations with river ecology is a key issue for policy makers charged with macrophyte control, and represents an important area of research (Baattrup-Pedersen & Riis 2004; Bal et al. 2011). Development and assessment of management strategies requires quantification of vegetation abundance, as well as data on the spatial distribution and sizes of macrophyte patches. However, the technical capability to do this accurately at a large scale has previously been limited (Nepf 2012). Established techniques such as waded cross-sectional surveys (Riis et al. 2003; Riis et al. 2008; Gunn et al. 2010) or sonar (Ibáñez et al. 2012b; Tena et al. 2017) provide estimates of cross-sectional blockage by macrophytes, but they are less suitable for evaluating patch
size characteristics, surface area blockage factor and spatial distributions of macrophyte patches. These techniques are also time consuming when applied at the reach scale, assume measurements of vegetation cover and river geometry are representative of regions between measurement locations, and can result in data with limited precision. For example, Ibáñez et al. (2012b) reported macrophyte cover increasing from 36.59% to 55.85% two days after a flushing flow, which is likely a reflection of the measurement uncertainty rather than a real change.

An alternative approach to ground-based reach surveys is the use of aerial photography. This approach is well suited to map submerged aquatic vegetation (SAV) in shallow rivers or lakes when turbidity is low (Visser et al. 2013; Verschoren et al. 2017). In the past, aerial images were obtained from surveys using light aircraft or helicopters (Marshall & Lee 1994), with the associated high cost of acquiring the data and decreased pixel resolution due to aircraft altitude. The recent availability of low-cost Unmanned Aerial Vehicles (UAVs) or ‘drones’ (Watts 2012) has enabled rapid and inexpensive acquisition of aerial images of SAV (Husson et al. 2014; Flynn and Chapra 2014). These high-resolution, low-altitude images improve classification accuracy and feature extraction (Rivas-Casado et al. 2016), as well as facilitating reconstruction of river morphology using Structure-from-Motion (SfM) techniques (Westoby et al. 2012; Marteau et al. 2016). Further information for feature extraction and delineation of object boundaries can be obtained using multispectral or hyperspectral imagery (Williams et al. 2003; Hestir et al. 2008; Silva et al. 2008). However, hyperspectral cameras are currently expensive, heavier than normal cameras, and may be sensitive to vibration and orientation due to long ‘pushbroom’ or ‘whiskbroom’ image capture time (Ortenberg 2016). Thus,
while hyperspectral technology provides exciting aerial imagery prospects, it is currently impractical for routine monitoring with consumer grade UAVs.

Once aerial images are acquired and processed into geo-referenced orthophotos, classification of features of interest (i.e. vegetation) can either be performed manually (Husson et al. 2014) or automatically (Flynn and Chapra 2014; Rivas-Casado et al. 2015; Visser et al. 2016; Senthilnath et al. 2017). Automatic classification methods are useful for estimating parameters such as total vegetation cover; however, it is challenging to delineate between neighbouring patches with touching boundaries, a feature that is crucial for evaluating the size of individual plants. Also, many automatic classification approaches require the development of large rule sets (Visser et al. 2016), which is time consuming, and it remains unclear whether classification rule sets generalise to other river reaches. Depending on the application, it may be more efficient and accurate to perform manual classification (Husson et al. 2016).

Although remote sensing of vegetation using UAVs is becoming standard practice, many studies focus on the utility of the aerial surveying method itself (Flynn and Chapra 2014), rather than its ecohydraulics applications. Survey data can, for example, be used to assess size distributions of macrophytes, their spatial arrangements, flow directions in patch mosaics, or coupled with measurements of hydraulic conditions to investigate macrophyte habitat use. Such field data, especially on size distributions, is critical for designing realistic laboratory or flume experiments on flow-vegetation interactions. However, available data are typically limited to simple bulk statistics, such as mean length (Spink 1992; Poynter 2014) or, most commonly, percentage cover (i.e. Riis & Biggs 2003; Gurnell et al. 2010; JNCC 2011; Pattison et al. 2017). This lack of
data, even for common macrophytes such as *Ranunculus* spp., creates uncertainties when designing experiments (Siniscalchi *et al.* 2010) or in physical modelling where vegetation-specific scaling relations are required (Thomas *et al.* 2014). Previous studies have established that SAV abundance is a function of hydraulic conditions and substrate stability (Butcher 1933; Riis & Biggs 2003; Franklin *et al.* 2008; Gurnell *et al.* 2010). However, there remains a paucity of data on hydraulic habitat use by vegetation at the sub-reach scale (Hart *et al.* 2013). There is also a lack of data on how dense regions of vegetation (clusters of patches) can influence flow patterns at the patch mosaic scale.

To reduce the knowledge gaps outlined above, this paper couples aerial assessment of the size characteristics of *Ranunculus penicillatus* patches with hydraulic data collected using an Acoustic Doppler Current Profiler (ADCP) and a piezometer device. The specific objectives were to: (i) evaluate the utility of UAV remote sensing to obtain macrophyte patch sizes and cover at the river reach scale; (ii) assess the geometry and size distributions of patches; iii) investigate the relationships between macrophyte cover, patch sizes and the hydraulic conditions of the habitats that they utilise; and iv) investigate the effect of vegetation clusters on flow at the patch mosaic scale.

2. Materials and methods

2.1. Study reaches

Fieldwork was conducted in the River Urie (North East Scotland), near the town of Inverurie (Figure 1). Aerial images and hydraulic data (velocity profiles, depth and water surface slope) were collected for two study reaches, which were separated by a section of river containing a bridge and overhanging trees (where safe UAV flights were not
possible). Reach 1 was a straight section (sinuosity index = 1.002, following Dey (2014)) with moderate macrophyte resistance (Manning’s $n = 0.049 \text{ m}^{-1/3}\text{s}$, following Graf & Altinakar (1998)). Reach 2 was a meandering channel consisting of three adjacent bends (a, b, c) defined between centreline inflection points (Figure 1) (Sinuosity index = 1.134, 1.366, 1.183). Reach 2 was densely vegetated with macrophytes, which was reflected by a fairly high Manning’s $n$ of 0.113 m$^{-1/3}\text{s}$.

### 2.2. Field measurements and background hydraulic conditions

Field data comprised of geo-referenced UAV images, with complementary hydraulic measurements from an ADCP and a piezometer (Table 1). Fieldwork days were selected based on favourable weather and stable river conditions. Ground Control Points (GCPs) were installed and surveyed on the 22nd of October 2015, with UAV flights and ADCP measurements conducted the following day. Piezometric slope measurements were taken on the 1st of November 2015, with the delay between the second and third measurement days due to rainfall on the 24th of October, which increased river discharge and delayed collection of comparable hydraulic data. River stage was stable during both days of hydraulic measurements (23rd Oct stage 45.1-45.2 cm; 1st Nov stage 46.8-46.5 cm) and sufficiently similar to allow use of piezometer and ADCP data. An additional section of channel directly upstream from reach 2 (Figure 1) was included in the UAV flights and analysis of macrophyte geometry, but no hydraulic data were collected there due to the rainfall event on the 24th October. Reach lengths and areas were estimated using ESRI’s ArcGIS GIS software from the final geo-referenced orthophotos (Table 1).

Discharge and river stage data (at 15 min. intervals) were provided by the Scottish Environment Protection Agency (SEPA) from their Pitcaple gauging station, ~14 km
upstream from the study reaches following the river centreline (~8.5 km point-to-point). Between the gauging station and the study reaches there are no major tributaries, so when river stage was stable at Pitcaple it was considered sufficiently stable at the study reaches to allow hydraulic measurements.

Hydraulic data were collected to be representative of conditions during the macrophyte growing season. The median discharge at the Pitcaple gauging station during the UAV and ADCP surveys was 1.62 m$^3$s$^{-1}$, while the median discharge during the piezometric water slope measurements was 1.87 m$^3$s$^{-1}$. The median discharge during the growing season (April to November) was 1.68 m$^3$s$^{-1}$, so the hydraulic measurements during the ADCP surveys were representative of summer growing conditions.

2.3. Recording and processing aerial images

Aerial images were acquired from a DJI Phantom 1 UAV with a 12MP GoPro Hero 3+ Black edition camera. The camera was mounted on a Zenmuse H3-2D Gimbal to maintain vertical orientation and maximise stability. To record the images, the UAV was flown along the general centreline of the river channel, with the altitude (approximately 20 m) being a balance between visibility of Ground Control Points (more GCPs visible with increasing height) and the macrophytes (less clear with increasing altitude). GCPs consisted of laminated A4 sheets, displaying a black cross, which were placed along each bank at approximately 20 m intervals. Terrestrial vegetation was cleared around the GCPs to maximise aerial visibility. Each GCP was geo-referenced using a Real Time Kinematic (RTK) Global Positioning System (GPS) which had a measurement error of 2.4 cm or less (Marteau et al. 2016). Aerial images were taken in timelapse mode, at one second intervals, with an image size of 4000x3000 pixels. They were manually screened for
clarity and degree of overlap, then the number of images was reduced to approximately one image per metre of river centerline (Table 1) before further processing. UAV flights were conducted around solar noon to maximise sun elevation for optimal macrophyte illumination, with minimal bank shadows and surface reflections. GCP installation, geo-referencing and UAV flights required a team of 3 and took less than one day.

The GoPro camera had a wide angle (‘fish-eye’) lens necessitating image correction. This was performed in the GNU Image Manipulation Program (GIMP), with the additional Batch Image Manipulation Plugin (BIMP) for multiple file processing, and ‘plug-in-lens-distortion’ for de-warping. After de-warping, the images were stitched using Agisoft Photoscan Professional and then output as orthophotos for each reach. Output orthophotos had a ground sample distance of 1 cm/pixel or better. Orthophotos were geo-referenced in the software QGIS using the ‘Geo-referencer’ plugin. The geo-correction transformation used was the ‘Thin Plate Spline’ approach, which fits GCPs exactly and allows for some localised image warping. This method was used since images were stitched along the centreline of the river only (Figure 1) with no side overlap compared to the flight paths used when mapping terrestrial areas. Nearest neighbour resampling was used to avoid blurring and smoothing which occurs when using an interpolation approach. More recent versions of Agisoft Photoscan Professional incorporate a georeferencing module, which is preferable to georeferencing in external software. The final geo-referenced orthophotos were then checked for scale accuracy by evaluating the length and area of known objects (the A4 ground control markers, storage bins and the ADCP boat) in ArcGIS. The correspondence of the measured area to the actual area of objects was good throughout the study reaches (difference between measured area and real area was within ±2% in all cases, mean error -0.153%, STD
0.582%, for 18 measurements of objects with known dimensions distributed throughout the study reaches). Image ghosting (moving subjects in original images being visible at multiple locations in stitched orthophotos) was investigated by comparing stitched orthophotos with the original images. Ghosting was occasionally observed for objects with very large displacement (such as the UAV pilot), but extraction of moving macrophytes was found to correspond to single photos and significant ghosting was not observed. Processing of geo-referenced orthophotos was performed in Matlab, with manual image classification by drawing polygons to delineate macrophyte patches. Manual rather than automatic classification was used since automatic classification algorithms did not reliably separate neighbouring macrophyte patches. Manual delineation of macrophyte boundaries required approximately 2 days. Macrophyte geometric parameters were geo-referenced to the centre of mass of each plant.

2.4. Hydraulic data

Depth, velocity profiles and discharge were determined from cross-section transects with a Teledyne StreamPro ADCP. Cross-sections were located at approximately 40 m spacing along the channel centreline, with 0.5 m lateral spacing in reach 1 and 1 m in reach 2 (due to time constraints). There were 4 cross-sections in reach 1 and 11 cross sections in reach 2 (Table 1). StreamPro ADCPs measure depth and fluid velocity (all three components of the velocity vector) in a vertical profile of ‘bins’. They are commonly used in flow gauging applications (Mueller & Wagner 2009), where the instrument is slowly moved from one bank of the river to the other. To use an ADCP for this purpose an internal feature called ‘bottom tracking’ locates the instrument in space by dead-reckoning. This information is then used in conjunction with the velocity bins and depth to estimate total discharge. However, the ‘bottom tracking’ feature only works when the bed is stable. As
such, it is not reliable in vegetated rivers or during flood events when the bed is moving. These problems can be circumvented by geo-referencing the ADCP position manually, or using a boat mounted RTK-GPS receiver. In the River Urie problems caused by vegetation movement were mitigated by geo-referencing the ADCP manually (distance across survey transects relative to GCPs measured using a tape measure). At each measurement ‘point’ (i.e. vertical profile) across the transect, data were collected over 30 seconds with 5 cm vertical bin spacing. This allowed surveys to be completed rapidly (within one day), thus limiting changes in river stage and discharge. ADCP positioning and stability were achieved using a four line configuration where substantial tension was applied to the bottom two lines to maintain orientation perpendicular to the cross-section (yaw), while roll and pitch were reduced by the top two lines. The contribution of translational motion to errors in velocity estimates was greatly reduced by temporal averaging, since the contribution of boat translational errors to mean velocity errors will tend to zero as measurement duration increases. This was the approach followed in all ADCP measurements, where 30 second profiles were considered a suitable tradeoff between spatial coverage and error minimisation.

ADCP velocity data were extracted in local XYZ coordinates and then converted to East, North, Up (ENU) coordinates by applying a rotation matrix, where the rotation angle was derived from the cross-section GCPs and orthogonality of the local XYZ system. In cross-sections free from vegetation, the velocities in blanking regions (near the bed and surface) were estimated by fitting then extrapolating a logarithmic profile to the velocity data. These velocity profiles were then vertically averaged and used with ADCP depth data to estimate bin discharges and total discharge. In densely vegetated cross-sections, logarithmic profiles did not apply, as flow was highly three-dimensional.
In these situations, depth-averaged velocity was estimated from valid data bins and no extrapolation was performed. This introduced uncertainty, with data from these cross-sections only used for flow visualisation and not estimates of the discharge at the study site, which was computed as the average of 6 discharge estimates from non-vegetated cross-sections. The average discharge at the study site during the UAV and ADCP surveys was 2.08 m$^3$s$^{-1}$, compared to the discharge of 1.62 m$^3$s$^{-1}$ at the Pitcaple gauging station which is approximately ~14 km upstream. Cross sectional mean velocities were estimated from discharge and cross-sectional area, which was obtained from trapezoidal integration of depth profiles. Depth in vegetated regions can be obtained from ADCP data more reliably than velocity, since oscillating macrophyte patch ‘tails’ intermittently expose the river bed. When the bed was hidden by the ‘head’ of a macrophyte patch, depth was approximated by interpolating between neighbouring profiles.

A piezometer device was used to determine water surface slope at approximately 20 m spacing along the channel centreline using the principle of hydrostatic levelling (Gordon et al., 2004). A piezometer provides equivalent water surface slope to a manometer, but has the operational difference that the upstream end of a piezometer is submerged to include hydrostatic pressure head, which avoids problems leveling the upstream end of a manometer with the water surface. Care must be taken that the inlet of a piezometer is placed in a quiescent area (such as within riparian vegetation that hang into the river) to avoid velocity head. The piezometer device had a 50 m hose, where the selection of hose length was a tradeoff between measurement accuracy (longer hose) and improved spatial resolution (shorter hose). Each piezometer measurement was repeated at least three times, with mean values then used. The spatial reference for piezometer
measurements was taken to be the mid-point of the hose, with measurements coincident with the geo-referenced cross-sections.

2.5. *Macrophyte geometry, cover and orientation*

Macrophyte patch geometry was characterised using the following descriptors: (i) planform area, defined as the sum of macrophyte pixels multiplied by the area of a single pixel; (ii) length, defined as the maximum separation of any two pixels belonging to the same macrophyte patch; (iii) patch orientation, defined as a unit vector originating at the patch centre of mass and parallel with the longest patch axis; (iv) patch width, defined as the maximum width orthogonal to the patch orientation unit vector (this definition provides a more robust estimate than projected width, which can result in overestimation of width for macrophytes with C shaped geometry caused by macrophytes aligning with flow around obstructions); and (v) patch aspect ratio, defined as length divided by width. Macrophyte perimeter was not used since it is dependent on the measurement resolution (similar to the coastline paradox of Mandelbrot (1982)).

Macrophyte cover was evaluated in each study reach with 10 cm centreline spacing, and was defined as: \( PMC(l) = \frac{\sum_{m=1}^{M} b_m(l)}{b_R(l)} \) where \( PMC \) is ‘proportional macrophyte cover’, \( l \) is the natural streamwise coordinate, \( b_R \) is the river width, \( b_m \) is the patch span orthogonal to the centreline curve at the cross-section, \( m \) is the macrophyte patch index across the cross-section and \( M \) is the total number of macrophyte patches that are present in the cross-section. Proportional macrophyte cover is analogous to surface area blockage factor \( B_{SA} \) (Green 2005), with zero streamwise averaging length, such that width instead of area is used to avoid smoothing macrophyte cover. When evaluating proportional macrophyte cover at the few locations with overhanging
vegetation, $b_R$ was replaced with $b_{RL}$, which is the width of the river section that is subject to incident light. Use of $b_{RL}$ helps avoid spurious relationships between macrophyte distributions and hydraulic characteristics that may occur due to light conditions and macrophyte visibility in the aerial images.

Macrophyte orientation was evaluated from the angle between the macrophyte unit vector and the river centreline unit vector (which represents, to a certain degree, the direction of ‘bulk’ velocity). Macrophyte orientation was then presented as a frequency distribution of the orientation angle.

2.6. Hydraulic habitat conditions

Hydraulic habitat conditions were characterised using the following standard descriptors: (i) cross-sectional mean velocity $U$, where $U = Q/A$, $Q$ is water discharge and $A$ is cross-sectional area; (ii) cross-sectional mean depth $D$, where $D = A/b_R$; (iii) Froude number $Fr$, where $Fr = U/\sqrt{gD}$ and $g$ is gravitational acceleration; (iv) water surface (piezometric) slope $S_p$, where $S_p = \sin(|\theta|)$ and $\theta$ is water surface angle (water surface slope and energy slope were found to be approximately equal for the cross-section spacing in our study reaches); (v) stream power per unit area $SP_A$ (Bagnold 1966), where $SP_A = \rho g Q S_p/b_R$ and $\rho$ is water density; and (vi) stream power per unit volume $SP_V$, where $SP_V = \rho g Q S_p/A = \rho g S_p U$. Stream power per unit volume is a useful hydraulic parameter for flows in highly vegetated reaches, where flow resistance (drag) acts throughout the water column and not just at the channel boundaries.
To estimate stream power, information on river slope is needed. Piezometers are able to determine water surface elevation changes with millimetre accuracy. However, they introduce significant spatial averaging along the channel centreline, due to the length of the piezometer tube. To account for this spatial averaging, it is appropriate to average macrophyte cover over the equivalent length before comparison. This can be achieved using the surface area blockage factor $B_{SA}$ for the channel segment corresponding to the length of the piezometer tube (50 m in this application) or through spatial averaging of proportional macrophyte cover. Which approach is optimal depends on the application and how weightings should be apportioned. Using $B_{SA}$ will apply higher weighting to wider cross-sections, since they contain more data, while spatial averaging of proportional cover will maintain every cross-section with the same weighting. In the present case, the spatial averaging of proportional macrophyte cover was used, since it allowed direct comparison with other relationships evaluated at finer spatial resolution (0.1 m cross-section spacing). The one-dimensional spatial averaging was performed with a box averaging filter corresponding to a centreline length of 50 m. The averaging filter was truncated at the ends of the measurement reaches, to avoid introducing error from padding with zeros, or assuming any macrophyte distribution beyond the known data. For example, the averaging filter shrinks from 25 m upstream and 25 m downstream to 25 m in only one direction as it approaches either end of the river reach (with filter normalisation adjusted as its length changes). This approach then allowed evaluation of smoothed proportional macrophyte cover as a function of those hydraulic parameters that involve water surface slope.

2.7. Flow-vegetation interactions at the patch mosaic scale

Maps were produced in QGIS to help assess how dense regions of vegetation affect local
flow conditions. The orthophotos of the study reaches provided the base layer, with geo-referenced macrophyte area, patch orientation, patch aspect ratio and ADCP measured velocities providing additional layers of data. The macrophyte patch orientation data indicated local flow direction in the patch mosaics while patch aspect ratio may serve as a measure of relative flow velocities, since flexible macrophytes are more streamlined in regions of higher fluid velocity (O’Hare et al. 2007). Macrophyte area was used to manually identify regions of high vegetation density (vegetation clusters).

3. Results

3.1. Utility of UAV remote sensing to obtain macrophyte patch sizes and cover

UAV remote sensing proved to be a practical approach for mapping aquatic vegetation. The surveys, conducted during summer median flows, yielded information on 1,099 R. penicillatus patches. Geo-referenced patch geometry data was easily extracted from the survey orthophotos and enabled coupling with field hydraulic data to investigate vegetation habitat utilisation. Geo-referenced data on macrophyte patch area, orientation, and aspect ratio provided a useful visualisation of local flow patterns and identification of dense macrophyte clusters, as detailed further below.

3.2. Vegetation geometric properties

The geometric properties of the 1,099 macrophytes are summarised in Table 2, while frequency distributions of individual parameters follow in Figure 2. The positive kurtosis values in Table 2 show that the distributions are more spread out than a Gaussian (tail
heavy), while positive skewness indicates that most distributions (except patch orientation) are asymmetric with tails of high positive values. A frequency distribution of patch planform area is shown in Figure 2a. The median planform area was 1.10 m² (Table 2), with 90% of the macrophytes having an area between 0.22 m² and 3.24 m². Only 1.91% of macrophyte patches had an area greater than 4 m². A frequency distribution of patch length is shown in Figure 2b. The median length was 2.77 m (Table 2), with 90% of patches having a length between 1.22 and 5.37 m and 2.18% being over 6 m long. A frequency distribution of macrophyte patch aspect ratio is shown in Figure 2c. The median patch aspect ratio was 5.13 (Table 2), with 90% of patches having an aspect ratio between 2.67 and 10.63, and 0.55% over 15. Macrophyte patch width generally increased with length, and the relationship was not dependent on bank proximity (Figure 3a).

A frequency distribution of macrophyte patch angle to the channel centreline is shown in Figure 2d. The orientation angle of 90% of the macrophyte patches was between -17.53° and 24.25°, with only 1.00% having angles outside ±40°. The mean patch angle was 0.74° (Table 2), which indicates that average macrophyte orientation was dominated by the bulk velocity. Macrophyte patch angle to the river centreline was not a function of proximity to the centreline or banks (Figure 3b) and roughly indicated that larger patches were more likely to follow the direction of bulk velocity.

**3.3. Coupling vegetation abundance and hydraulic conditions**

Relatively few plants were located in regions of low cross-sectional mean velocity (Figure 4). A Gaussian function provided a suitable fit to the data and offered a physically valid extrapolation beyond the range of recorded cross-sectional mean velocity. Macrophytes were more abundant in riffles than pools, which is reflected by their utilisation of habitats
with higher cross-sectional mean velocities. The dependence of macrophyte cover on both velocity and depth was evaluated using the Froude number (Figure 5). Macrophyte habitat utilisation indicated a bell-shaped dependence on the Froude number, with an improved fit compared to flow velocity or depth alone.

Macrophyte cover in the study reaches was also related to stream power per unit area (Figure 6) and stream power per unit volume (Figure 7). Both indicated that habitat utilisation followed a bell-shaped function. There was no clear relationship between the planform areas of macrophyte patches and stream power per unit area (linear regression, \( p = 0.83, R^2 = 0.0042 \); non-linear regression with a Gaussian function, \( R^2 = 0.24 \)), but the density of macrophytes per unit bed area clearly was a function of stream power per unit area (Figure 8; non-linear regression with a Gaussian function, \( R^2 = 0.89 \)). This difference indicates that stream power influences vegetation cover via effects on the number rather than the size of patches.

3.4. The effect of vegetation clusters on flow at the patch mosaic scale

Flow was diverted around dense vegetation clusters (Figure 9), with sedimentation observed downstream of the clusters, and erosion observed where flow was redirected towards the outer banks. Flow in the macrophyte patch mosaics showed high spatial heterogeneity and cross-stream mean velocities should not be assumed to be zero.

4. Discussion

4.1. Utility of UAV remote sensing to determine macrophyte patch sizes and cover

UAV remote sensing allowed an unprecedented 1,099 \( R. penicillatus \) patches to be
surveyed, with images analysed to determine patch 2D geometry. The study reaches were surveyed under flow conditions typical of those prevailing in the summer growing season. Aerial data were coupled with in-stream hydraulic measurements to assess distributions of *R. penicillatus* in relation to flow hydraulics at typical (i.e. close to average) summer river discharges. At the end of summer (when surveys were conducted) *R. penicillatus* patches had large planform areas (Figure 2a), presumably to intercept light and vertical fluxes of nutrient-containing water. These natural patches are 1-3 orders of magnitude larger than those used in laboratory experiments such as Siniscalchi et al. (2010) and Ortiz et al. (2013). This difference emphasises that some of the larger scale phenomena observed in field settings may not occur in the laboratory. Future laboratory experiments on flow-vegetation interactions could usefully incorporate the information on *R. penicillatus* sizes provided here, to generate experimental designs with macrophyte patch size and bulk velocity scaled to better represent natural macrophytes and field conditions.

### 4.2. Geometry and size distributions of macrophyte patches

Macrophyte lengths (Figure 2b) indicated a log-normal distribution (with arithmetic mean $2.84 < \mu_r < 3.11$, and arithmetic standard deviation $1.25 < \sigma_r < 1.54$ for 99% confidence levels). This may be due to a normal distribution of patch sizes at the start of the growing season, then differential rather than linear growth (Huxley 1932) (i.e. the patch growth rate being proportional to the patch size itself rather than independent of it). Such differential growth could occur if plant stems continue to grow along their entire length, or if nutrient and energy collection from the entire organism enhances growth rates in certain regions. Alternatively, the differences in macrophyte lengths could be a function
of habitat suitability, related for example to hydraulic constraints on growth or differences in light and nutrient availability.

Macrophyte aspect ratio (Figure 2c) indicates the importance of streamlining to reduce the total drag imposed by flowing water. However, the observed streamlining is likely to be a tradeoff with the growth rate, since planform area for light interception is reduced. Further studies that account for water velocity around each patch and vegetation biomass are needed to determine if this evolutionary trade-off is standardised across macrophyte patches. Macrophyte length, width and aspect ratio was not dependent on channel bank proximity (Figure 3a). This indicates that shading from the banks and hydraulic differences across the channel span, are not as important to plant characteristics as changes which occur along the channel centreline (e.g. channel slope, mean velocity, pool riffle sequences). Macrophyte width generally increased with length (Figure 3a), although data on local flow fields around macrophytes is needed to assess the influence of local drag and reconfiguration on their aspect ratio.

The mean macrophyte patch angle of 0.737° (Table 2, Figure 2d) indicates no net bias in macrophyte orientation and shows that orientation is dominated by the direction of the bulk velocity. There were also few outliers of patches inclined at large angles to the river centreline. However, the standard deviation of 13.0° indicates that localised variation of flow direction is significant and that cross-stream mean velocities in patch mosaics should not be approximated as zero, even near the banks (Figure 3b).

4.3. Hydraulic habitat utilisation by *R. penicillatus*

The relationship between proportional macrophyte cover and cross-sectional mean
velocity (Figure 4) suggests that specific patterns of habitat utilisation for *R. penicillatus* develop during the growing season. Although high flows can occur during the summer, sustained period of relatively low, stable flow tended to dominate the River Urie hydrograph in 2015. At a discharge of 1.62 m$^3$s$^{-1}$, the highest cover values of *R. penicillatus* in the study reaches occurred in locations where velocity was around 0.4 ms$^{-1}$. This agrees with the results of Riis & Biggs (2003) and Poynter (2014), both of whom reported maximum *Ranunculus* spp. cover between 0.3 and 0.5 ms$^{-1}$.

It has been suggested that macrophytes serve as ecosystem engineers (e.g. Gurnell 2014) that will reduce mean river velocities through hydraulic resistance as their abundance increases. Thus, it is very likely that the sections of a river reach with mean velocities of around 0.4 ms$^{-1}$ and maximum vegetation cover would originally have had higher flow velocities. These velocities would then systematically decrease and water level would increase as macrophyte abundance and hydraulic resistance increased. This process could continue until maximum macrophyte cover and velocities of around 0.4 ms$^{-1}$ are reached, at which point the system may become stable (or in equilibrium) with rates of biomass accumulation balancing rates of removal during intermittent scouring floods. The upper bounds of the utilisation curve in Figure 4 may be due to hydraulic control of macrophytes through drag and erosion. However, further studies in catchments with higher prevailing summer velocities would be needed to test this hypothesis and determine if a bell-shaped (Gaussian-type) utilisation curve for velocity is a universal trait of *R. penicillatus*.

The lower bounds of the velocity utilisation curve (Figure 4) may be due to depth, since cross-sectional mean velocity decreases when cross-sectional mean depth increases
(if river width is constant). The data supports this hypothesis, with decreasing macrophyte cover as depth increases. There are several potential physical mechanisms to explain this effect. The first is that as depth increases, light availability decreases, which limits macrophyte growth and abundance. The second is that as depth increases, vertical gradients of streamwise velocity and turbulent stresses decrease; this reduces bed shear stress and results in an accumulation of fine sediment. This fine sediment may then smother macrophytes, or provide weak anchorage for their roots, resulting in higher removal rates during flood events. The third explanation is that these deep regions may experience decreased colonisation, since vegetation is positively buoyant and there may be insufficient rough obstacles to trap propagules (personal communication with Dr. Matthew O’Hare). However, another unknown mechanism such as increased herbivory by grazers (Carreira 2014) that can be more abundant in pools (Englund & Krupa 2000) is also possible. Of these, the dominant control may be flow-vegetation-sediment interactions, since they can also explain the evolution of vegetation patch mosaics. For example, sedimentation in seagrass stems and cylinder arrays is dependent on canopy density (Bouma 2007; Lawson et al. 2011; Follett & Nepf 2012), where erosion is reported for low canopy density and deposition for high canopy density. The same principle may also apply if the scale is increased, such that each macrophyte patch represents a single roughness element, with the density of ‘macrophyte clusters’ determining net erosion or deposition. In dense macrophyte clusters, deposition of fine sediment would be enhanced, with accumulation near the centre in regions of low fluid stress. This accumulation of fine sediment may decrease rooting strength, leading to hydraulic removal of the cluster centre during the next flood event. This mechanism would return the geometry to a patch mosaic, rather than complete bed cover, generating a relatively stable, self-regulating system.
Froude number (Figure 5) provided an improved prediction of macrophyte abundance over the cross-sectional mean velocity alone (Figure 4), which supports the hypothesis that both depth and velocity are important factors influencing macrophyte abundance. Peak macrophyte abundance occurred for a Froude number around 0.21 and then declined at higher $Fr$ values. The peak may represent an optimum balance of nutrient flux, light availability, hydraulic stress and sediment composition (for anchoring strength and nutrient availability). Hydraulic conditions during measurements were close to the median summer growing conditions, limiting the range of $Fr$ values recorded. Although not the objective of the present study, it would be informative to survey abundance throughout the growing season to determine if flood events control the shape of the utilisation curve. Similarly, it would be informative to survey reaches where flow hydraulics at summer median discharges differ from those reported here, especially to determine whether cover values conform to a bell-shaped habitat utilisation curve with Froude numbers greater than 0.25.

*R. penicillatus* stream power relations (Figures 6 and 7) suggest that hydraulic stress on macrophytes and substrate through the dissipation of gravitational potential energy is an important predictor of macrophyte abundance. In this context, a relevant question is whether patch size, or patch number density is responsible for the changes in abundance across the range of stream power values recorded. The number density of patches in the River Urie was a function of stream power (Figure 8). However, the size of patches was not (regression of average macrophyte area against stream power per unit bed area yielded; a) linear fit $R^2=0.0066$ and $p=0.78$; b) second degree polynomial fit $R^2=0.24$; c) Gaussian fit $R^2=0.24$ and 95% confidence levels of $2.4<\mu<5.3$, $2.0<\sigma<8.3$).
This provides evidence that it is vegetation control at the patch mosaic scale, through the removal of entire patches, rather than ‘self-pruning’ of patches (de Langre 2008; Lopez et al. 2011), that governs reach scale macrophyte cover. This is supported by visual observations made at the study site during floods, when whole patches were recorded floating downstream, rather than plant fragments. The reason why stream power was a good predictor of vegetation number density (Figure 8) may be due to its effect on bed sediment composition, where moderate values could limit the buildup of fine sediment and provide a rougher, more stable, substrate for vegetation to attach to. High values of stream power could also provide an upper control on vegetation abundance through their effect on vegetation drag and sediment transport processes. At sufficiently high stream power, biomass would be removed by uprooting, or abrasion by mobile sediment (Hoyle et al. 2017).

4.4. The effect of vegetation clusters on flow at the patch mosaic scale

Dense clusters of macrophyte patches were easily identified from the air (Figure 9). These clusters can cause spatial heterogeneity in flow resistance, resulting in flow redirection around the cluster. Flow redirection towards outer banks may cause erosion and undercutting (Figure 9), while reduced flow and fluid stresses inside the cluster may result in localised sedimentation. Extensive areas of river channel can be covered with the aerial surveying and visualisation techniques reported here, providing a more efficient assessment than ground-based surveys. Thus, vegetation mapping and flow visualisation may be useful to assess areas at risk of bank erosion, channel sedimentation, or bank overtopping in floods due to channel blockage. Such information should enable better-informed river management.
5. Conclusions

Geometric information on 1,099 *Ranunculus penicillatus* macrophyte patches were obtained from aerial surveys and image analysis. This data on individual patches of *R. penicillatus* was long overdue with most other published work focusing on quantification of abundance at the reach scale (Wood et al. 2012) while the very few available studies of individual *R. penicillatus* patches (e.g. Spink (1992)) do not report distributions of size parameters.

At the end of the summer growing season *R. pencillatus* patches had mean planform area of 1.32 m$^2$, mean length of 2.95 m, mean aspect ratio of 5.63 and mean angle to the river centreline of 0.737°. Lengths followed a log-normal distribution, which was attributed to differential growth (Huxley 1932) (i.e. patch growth rate depended on the patch size itself). Macrophyte patch orientation data showed that on average plants followed the bulk velocity, although the standard deviation of 13.0° illustrated the substantial localised variation in orientation and spatially heterogeneous flow fields within patch mosaics. The size data in this paper will enable future experiments to be better scaled to reflect field conditions.

At typical summer (growing season) discharges, macrophyte abundance was related to river hydraulic conditions, with high vegetation cover corresponding to specific ranges of velocity, Froude number and stream power. This may be due to flow-vegetation-sediment interactions and hydraulic control through the removal of entire patches, since the number density of patches was a function of stream power while the average area of patches was not. Macrophyte clusters causing flow redirection, localised sedimentation and bank erosion were easily visible in the orthophotos. This information
could assist in targeted management of aquatic vegetation that avoids the ecologically destructive practice of clearing all river vegetation.

Acknowledgements

The authors would like to thank Matthew O’Hare, Jochen Aberle, Alexander Sukhodolov and Bernhard Statzner for valuable discussions and advice during the project. The authors would also like to thank Alasdair Matheson and the Scottish Environment Protection Agency (SEPA) for providing discharge and river stage data.

Disclosure statement

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

Funding

The work was part of the research project ‘Hydrodynamic Transport in Ecologically Critical Heterogeneous interfaces’ (HYTECH), the support of which, under the European Union's Seventh Framework Programme (Marie Curie FP7-PEOPLE-2012-ITN, European Commission grant agreement number 316546), is gratefully acknowledged. The work was also partially funded by the National Institute of Water and Atmospheric Research (NIWA) under the Sustainable Water Allocation Research Programme, the support of which is gratefully acknowledged.

References


Tables and Figures

Table 1. Summary of reach geometry, geographic data and hydraulic data.

<table>
<thead>
<tr>
<th>Site</th>
<th>Length (m)</th>
<th>Area (m²)</th>
<th>Number of aerial images stitched</th>
<th>Number of GCPs</th>
<th>Number of ADCP cross-sections</th>
<th>Number of local slope estimates</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reach 1</td>
<td>121</td>
<td>1,287</td>
<td>205</td>
<td>13</td>
<td>4</td>
<td>7</td>
</tr>
<tr>
<td>Reach 2</td>
<td>366</td>
<td>4,253</td>
<td>337</td>
<td>30</td>
<td>11</td>
<td>20</td>
</tr>
</tbody>
</table>

Table 2. Summary statistics for 1,099 macrophyte patches, with data quartiles from the cumulative distribution function (CDF). Information derived from the CDF: min = CDF(0%), median = CDF(50%), max = CDF(100%), and inter quartile range (IQR) = CDF(75%) - CDF(25%). Kurtosis is calculated as ‘excess kurtosis’ which is the fourth standardised moment minus three.

<table>
<thead>
<tr>
<th></th>
<th>Mean</th>
<th>Standard Deviation</th>
<th>Variance</th>
<th>Skewness</th>
<th>Kurtosis</th>
<th>CDF 0%</th>
<th>CDF 25%</th>
<th>CDF 50%</th>
<th>CDF 75%</th>
<th>CDF 100%</th>
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<tbody>
<tr>
<td>Area (m²)</td>
<td>1.323</td>
<td>0.9425</td>
<td>0.8883</td>
<td>1.271</td>
<td>1.848</td>
<td>0.04</td>
<td>0.61</td>
<td>1.10</td>
<td>1.78</td>
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<td>Length (m)</td>
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<td>1.259</td>
<td>1.586</td>
<td>1.023</td>
<td>1.916</td>
<td>0.31</td>
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<td>Width (m)</td>
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<td>0.3111</td>
<td>0.09680</td>
<td>0.9576</td>
<td>1.808</td>
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<td>0.45</td>
<td>0.63</td>
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<tr>
<td>Aspect ratio</td>
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<td>2.548</td>
<td>6.494</td>
<td>1.649</td>
<td>4.975</td>
<td>1.27</td>
<td>3.88</td>
<td>5.13</td>
<td>6.69</td>
<td>23.57</td>
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<tr>
<td>Orientation to centreline (degrees)</td>
<td>0.7370</td>
<td>13.01</td>
<td>169.2</td>
<td>0.3352</td>
<td>1.497</td>
<td>-67.63</td>
<td>-7.30</td>
<td>-0.36</td>
<td>7.65</td>
<td>50.24</td>
</tr>
</tbody>
</table>
Figure 1. Study reaches 1 and 2 in the River Urie (NE Scotland, UK), with geo-referenced orthophotos overlain onto satellite imagery of the area (courtesy of Bing maps and QGIS software). Sinuosity index = 1.002 (reach 1), 1.134 (reach 2a), 1.366 (reach 2b), 1.183 (reach 2c). Site coordinates N 57.277, E -2.365.

Figure 2. Frequency distributions of (a) planform area, (b) length, (c) aspect ratio, (d) macrophyte angle. Macrophyte angle is defined relative to the river centreline, where positive angles are macrophyte tails pointing towards the left bank.
Figure 3. (a) Macrophyte patch width as a function of length, (b) Macrophyte patch angle as a function of patch area. Macrophyte angle is defined relative to the river centreline, where positive angles are Macrophyte tails pointing towards the left bank. Dots are coloured by distance to the nearest bank divided by cross-section width.
Figure 4. Macrophyte cover as a function of cross-sectional mean velocity. Habitat utilisation indicates a normal distribution. Data combined from reach 1 and 2 of the River Urie.

Figure 5. Macrophyte cover as a function of Froude number. Habitat utilisation indicates a normal distribution. Data combined from reach 1 and 2 of the River Urie.
Figure 6. Macrophyte cover as a function of stream power per unit area. Habitat utilisation indicates a normal distribution. Data combined from reach 1 and 2 of the River Urie. Macrophyte cover smoothed over the length of the piezometer.

Figure 7. Macrophyte cover as a function of stream power per unit volume. Habitat utilisation indicates a normal distribution. Data combined from reach 1 and 2 of the River Urie. Macrophyte cover smoothed over the length of the piezometer.
Figure 8. Macrophytes per unit bed area vs stream power per unit area, evaluated over 50 m streamwise averaging domains. Data combined from reach 1 and 2 of the River Urie and indicates that the numerical density of macrophytes is a function of stream power. A normal distribution fit provides a possible extrapolation beyond the study site conditions.
Figure 9. *Ranunculus penicillatus* patches in the River Urie, showing (a) flow redirection around a dense macrophyte cluster that may have caused bank erosion and changes to river morphology, (b) spatial heterogeneity in vegetation biomass and hydraulic resistance at the sub-reach scale resulting in complex flow patterns, with implications for river morphology and localised sediment dynamics. Red arrows are patch orientation vectors scaled by aspect ratio, green circles show patch planform area.