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**Evidence of sedimentation inequality along riparian areas colonised by *Impatiens glandulifera* (Himalayan balsam)**

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**Summary**

Soil loss from riparian areas supporting the annual invasive weed, *Impatiens glandulifera* (Himalayan balsam), was measured and statistically compared to equivalent values recorded at nearby, topographically-similar areas supporting perennial vegetation over a cumulative seven-year period, along sections of two separate river systems; one in Switzerland, and one in the UK. Soil loss from colonised locations was significantly greater than from reference locations in four of the seven measurement periods. Despite contrasting results, standard deviations, based on soil losses and gains were predominantly higher for colonised areas at both rivers over most monitoring periods. These findings indicated that areas colonised by Himalayan balsam experience higher sediment flux in comparison with areas free of invasion. Here, we test those original interpretations by re-interrogating the datasets using a more robust analysis of inequality. Nine datasets were tested, five of which (i.e. 56%) showed that sediment flux was significantly greater at Himalayan balsam-invaded areas than at reference areas. Three datasets showed no significant difference in sediment flux between invaded or reference areas (33%), and one (11%) showed significantly higher sediment flux at reference areas. Most results uphold our original interpretations and support our hypothesis that hydrochory probably dictates where colonisation initially occurs, by depositing Himalayan balsam seeds in slack or depressional areas along river margins. Once Himalayan balsam becomes established and sufficient perennial vegetation is displaced, seasonal die-off and depleted vegetation cover may increase the risk that some areas will experience significantly higher sediment flux.

**Introduction**

*Impatiens glandulifera* Royle (common English name, Himalayan balsam, and hereafter abbreviated to HB) is an invasive annual weed that was introduced to the UK and Europe in the mid-19th Century and was noted for its invasive tendencies around the 1930s (Beerling & Perrins, 1993). It is now well established in 23 of the 28 European Union (EU) member states (CABI, 2017). Himalayan balsam rapidly establishes itself in most environments (Andrews *et al*., 2005; Dassonville *et al*., 2008; Tanner & Gange, 2013; Pattison *et al*., 2016; Ruckli *et al*., 2016), but as a ruderal plant, it favours damp, nutrient-rich soils that experience frequent disturbance. Consequently, river margins represent prime habitat. Once established, it forms dense, and almost monocultural stands that range from a few square metres up to several thousand square metres. Mature HB plants can each produce in excess of 2,500 seeds, each of which can be ejected up to 7 m from the parent (Chapman & Gray, 2012). Given the proximity of HB to flowing water, many seeds are thus propelled into the river and transported downstream (Pyŝek & Prach, 1995). As an annual plant, HB is intolerant of cold weather, however. During the first frosts of the year, whole stands collapse and die simultaneously(Skálová *et al*., 2011), which can increase the vulnerability of the underlying soil to erosion.

Given the invasive tendencies of HB (Tickner *et al*., 2001), much attention has been devoted to understanding its effect on biodiversity (e.g. Hejda & Pyŝek, 2006; Hulme & Bremner, 2006; Tanner & Gange, 2013; Pattison *et al*., 2016). Despite warnings linking HB to elevated erosion (Roblin, 1994), quantifying its influence on sediment dynamics has, by contrast, received limited attention. Given the acknowledged importance of fine-sediment as a vector for the transfer of nutrients and contaminants (Walling *et al*., 2003), any process that augments sediment flux in fluvial environments not only increases the suspended sediment load, but also promotes the sorption/desorption of sediment-bound contaminants, resulting in poor water quality, degraded habitat, and reduced biodiversity (Horowitz *et al*., 2012).

One of the first empirical investigations to determine the influence of HB on erosion was undertaken by Greenwood & Kuhn (2013), who measured soil loss from HB-invaded areas along the Ibach river in northwest Switzerland over the winter 2012/13, and compared findings against equivalent data recorded from nearby topographically similar areas free of HB plants. Whilst that study reported significantly higher erosion from HB-invaded areas, the results from one die-off to germination cycle were standalone and thus required corroborating over multiple die-off to germination cycles, and from different river catchments. These criteria were met when Greenwood *et al*. (2018) reported additional erosion data from along the same watercourse (The Ibach) over the winter periods 2013-16, along with erosion data from an HB-infested floodplain of the River Taw, southwest UK, over three winter-periods 2013-16.

However, comparison of erosion data from the above monitoring periods returned conflicting results, with significantly higher soil loss being recorded from HB-invaded areas in only two out of the four-years from the Ibach (2012/13 & 2013/14), and in two out of 3-years from the River Taw (2013/14 & 2014/15). Despite the generally comparable findings between river systems, yet inconsistent statistical findings, standard deviation values, based on net sediment losses and gains (i.e. defined as ‘sediment flux’), were used as a comparison by which the relative stability of sediment-related processes at HB-invaded and reference areas was judged. Standard deviation values were consistently higher at invaded areas for both river systems over most monitoring periods (except the Ibach, 2014/15 and the River Taw, 2013/14), which was tentatively interpreted as an indication of greater sediment flux at HB-invaded locations in comparison with spatially- and topographically-linked areas free of HB invasion. However, given the fact that notable numbers of invaded sites were necessarily abandoned after just one monitoring season, usually due to high-magnitude erosion and no reinvasion by HB the following spring (Greenwood *et al*., 2018), the possibility exists that some sedimentation datasets, particularly from invaded areas, are skewed. Furthermore, use of the standard deviation as a measure of variability is often not ideal, because values will increase as raw data-values increase (Taylor, 1961). To overcome this possibility, and to explore more thoroughly whether inequalities in sediment flux exist between HB-invaded and uninvaded areas, we interrogate the original datasets more rigorously by applying a more robust statistical approach involving an analysis of inequality (Damgaard & Weiner, 2000).

Further to the above, and in response to reports that HB modifies certain soil physicochemical properties that confer its invasiveness (e.g. Andrews *et al*., 2005; Dassonville *et al*., 2008; Tanner & Gange, 2013; Pattison *et al*., 2016; Ruckli *et al*., 2016), Greenwood et al. (2018) compared organic carbon (OC) content, phosphorus (P) concentration, and in-situ bulk density (BD) between soils from invaded and reference areas for the Ibach catchment only. Although there was no difference in P concentration or OC content between HB-invaded and reference areas, BD was significantly higher for the latter group (ref. 0.82 g cm-3 Vs. HB 0.71 g cm-3). Combined with findings in which the roots of HB plants trap very fine-sediment (Greenwood *et al*., 2018), a hypothesis was formulated in which sedimentation processes are more prevalent at HB-invaded areas, as indicated by predominantly higher standard deviation values, because regular inputs of fresh material remain loose and uncompacted and are less able to resist erosive forces during high river-flow. Resulting from those previous findings, therefore, we also assess the variability of data for the above properties, OC, P and BD using inequality analyses.

**Materials and Methods**

*The study sites*

Sedimentation data was recorded along sections of two separate river systems, one in northwest Switzerland (The Ibach) and the other (the River Taw) in southwest UK. A detailed description of the Ibach catchment is given in Greenwood & Kuhn (2013), and in Greenwood *et al.* (2018) for the River Taw floodplain on which sedimentation data were recorded.

Briefly described for both systems, The Ibach catchment is located in the Jura mountains, approximately 20 km south of the city of Basel, in northwest Switzerland (coordinates: 47o 25’09.80’’ N; 07o 34’24.19’’ E). The catchment extends to ca. 10 km2, and the watercourse is approximately 10 km long, and flows predominantly northwards until it confluences with the Birs River. With a base-flow depth of ca. 0.11 m and channel widths of ca. 3.3-5.5 m, base-flow is estimated at 0.25 m3 s-1. Despite its short length and now geomorphologically underfit dimensions, the gravel-bed watercourse has incised through limestone bedrock to form steep-sided valleys with narrow (max. ca. 25 m wide) floodplains. Most valley-sides and river corridor are thickly forested, with permanent pasture accounting for ca. 10% of land immediately adjacent to the watercourse. Mean annual precipitation is ca. 850 mm, and peak rainfall at ca. 90 mm per month usually occurs from May to September. The valley-bottom has a mean elevation of ca. 500 m asl. Precipitation during winter months often falls as snow. Vegetation along the riparian zone consists predominantly of deciduous trees, the most common of which is *Fagus sylvatica* L (beech), but are interspersed with infrequent stands of conifers (*Pinus sylvestris L*). Upper valley-sides are largely devoid of understory vegetation, but lower valley-sides and valley-floors consist of a dense and unbroken understory of lower and higher-order plants; the most abundant of which (identified to genus level only) include *Allium* spp., *Hedera* spp. *Ranunculus* spp., *Rubus* spp., *Trifolium* spp., and *Urtica* spp. Himalayan balsam is common within the riparian zone but is particularly concentrated along river margins. Stands of HB are densest and most prevalent within the downstream half of the catchment. Overbank flood events are uncommon within this catchment due to the hydrology/precipitation regime, the small catchment size, and lack of upstream urban development; all of which curtail runoff generation.

The River Taw is located in southwest UK and is ca. 82 km long from source to outlet. With a catchment area of ca. 1,227 km2, the Taw drains the northeast and southern flanks of Dartmoor and Exmoor National Parks, respectively. The study site is situated on a ca. 1.0 km long floodplain reach, located on the west side of the river (coordinates: 50o 56’38.12’’ N; 03o 54’31.73’’ W), and approximately 40 km south east of the estuary outlet into the Bristol Channel. Base-flow is estimated at 6.0 m3 s-1, mean flow-depth during base-flow conditions is ca. 0.5 m, at which point, channel width at the chosen floodplain reach is estimated at between 12-15 m. The catchment area upstream of the study site is ca. 700 km2, the elevation of the floodplain reach is ca. 41 m asl, and is under permanent pasture. Annual precipitation within the Taw catchment is 1,180 mm and peak rainfall, at 120-140 mm per month usually occurs in autumn/winter period (i.e. October to November). As peak precipitation corresponds to the time when HB undergoes seasonal die-off and live HB plants are absent, this is assumed to render invaded sections of riverbank most vulnerable to erosion. Aside from stands of HB, vegetation along the river margins (identified to genus level only) mainly consist of *Galium* spp., *Rubus* spp., *Rumex* spp. and *Urtica* spp., but are interspersed with many (unidentified) grass species. Small and isolated copses of immature trees also grow along the river margins, and these, again identified to genus level only, include *Betula* spp., *Castanea* spp*., Crataegus* spp., *Fraxinus* spp. and *Quercus* spp. Himalayan balsam is prevalent along river margins on the western side of the floodplain, but infestation increases in both extent and density from south to north. In contrast with the Ibach, overbank flood events are common at this particular floodplain, and occur on average seven times per year, typically between October to March. During even a relatively low-magnitude flood event, inundation depths along downstream sections of the floodplain, where HB is concentrated, frequently exceed 1.0-2.0 m depth (Greenwood *et al*., 2013). The geometry and topography of this floodplain also ensures that overbank floodwaters remain well connected to the river and continue to be fast-flowing. Consequently, the timing of floods is relevant at this site, as any that occur soon after die-off tend to rip out dead and decaying remains of HB plants, often leaving affected land-surface areas completely devoid of vegetation cover.

*Measuring sedimentation*

Sites for erosion monitoring were selected based on the density of HB infestation (i.e. > 20 HB plants m-2), the relative accessibility of invaded sites, and whether topographically comparable reference areas could be identified close to an HB-invaded area. The close proximity of a potential reference site to an HB-invaded area also ensured that both locations shared similar surface hydrological conditions, which meant that each reference location provided an indication of background erosion rates at that location. A fuller description of the erosion monitoring equipment, their installation and measurement protocols are described in Greenwood & Kuhn (2013), and Greenwood *et al*. (2018). In brief, pairs of 6 mm dia. metal erosion pins, with lengths tailored to suit local soil-profile conditions, were driven into the soil profile at a distance of 1.2 m apart from each other, to form a transect. Pins were cut, if required, so that ca. 5 cm of the top of each pin remained above-ground. Transects were installed in HB-colonised areas and, as alluded to above, each transect was paired with a second, reference transect installed in a nearby area (max. ≤ 4.0 m distance) that not only remained free of HB infestation but also supported an assemblage of perennial vegetation. When taking soil surface profile (SSP) measurements, a 1.2. m long lightweight yet extremely rigid aluminium profile-bridge could be rapidly fitted onto each pair of pins. Ten holes, each 7 mm dia. were drilled through the bridge (i.e. from top to bottom) at equidistant (0.1 m) intervals along the central 1.0 m portion. These acted as predetermined measurement points and ensured that SSP values were consistently recorded at the same positions along a transect during each re-measurement campaign. When taking measurements, a digital Vernier Caliper was aligned through each hole and extended to the soil surface below to estimate the distance from profile bridge to the underlying soil. Errors in measurement estimates were minimised by taking duplicate measurements and averaging the two values. Monitoring campaigns at both river systems usually commenced around mid-October when HB plants had undergone seasonal die-off and all vegetation cover was generally less dense. Monitoring was terminated around mid-April when vegetation regrowth made relocating pins prohibitively difficult. Individual re-measurement campaigns were generally conducted every 4-6 weeks between October to April.

Arising from the above methodology, we have now accrued a 4-year continuous sedimentation record along a 2.0 km section of the Ibach for the HB die-off to germination periods, 2012-16, along with a 3-year continuous sedimentation record for the HB die-off to germination periods, 2013-16, along a ca. 1.0 km long floodplain reach of the River Taw. In addition to those winter datasets, we also have a 2-year summer sedimentation record from the Taw for the years 2015 and 2016, which is based on sedimentation values recorded at the end of the previous monitoring campaign (i.e. April) and compared against equivalent values recorded at the beginning of the following campaign (i.e. October). Collectively from the two river systems, this research has now generated a total of 6,740 individual SSP measurements. Each of these measurements contribute to tracking the evolution of the SSP along respective 1.0 m long transects; some installed within stands of HB and some installed in nearby, topographically comparable reference locations, to provide an indication of both sediment losses (erosion) and gains (deposition).

*Soil physicochemical properties – Ibach catchment*

More detailed information about the soil sampling and laboratory protocols is given in Greenwood et al. (2018). In brief, in-situ BD was measured in 2016 along selected HB-invaded and reference sites using a 5 cm diameter \* 5 cm long metal ring (equivalent to 100 cm3 volume). Prior to sampling, litter and organics were cleared from the surface, and the ring was driven into the soil profile to the required depth, which ensured that material was consistently sampled from 0 to 5 cm deep. Each sample was bagged, marked and transferred to the laboratory where they were individually weighed (moist) to one decimal place, and oven-dried at 40o C. Once dry, each sample was reweighed (dry) and data were processed, to determine the in-situ density of each soil sample (expressed in g cm-3). In total, 60 individual soil samples were collected; 30 from HB-invaded areas, and 30 from paired reference areas (i.e. 30 paired-samples).

Phosphorus extraction followed the protocol described in Anon. (2001) and firstly involved sampling soil from 0-5 cm depth, which was, again, oven-dried at 40o C, but screened through a 63 µm sieve. Approximately 5 g of soil (< 63 µm fraction) per sample was placed in an Erlenmeyer flask containing 50 mL of nitric acid (HNO3) at a concentration of 2 mol/L. Flasks were placed in a water-bath at 100o C for 2 hrs. and covered. The liquid was then filtered through a < 2 µm membrane (Schleichel & Schüll 602 1/2 ehØ18.5 cm) and stored to await analysis. Analyses were performed on an Inductively Coupled Plasma-Optical Emission Spectrometer (ICP-OES) (AVIO 500, PerkinElmer USA). Standards were prepared at 1, 10 and 100-times dilution factors. Resultant data are expressed in milligrams per litre (i.e. parts per million (ppm)) and measurement uncertainty is generally ≤ ± 5%.

Organic carbon content was quantified using a Leco RC 612 multiphase carbon and hydrogen/moisture analyser (LECO Corporation USA). Sample preparation involved placing ca. 0.5 g of < 63 µm soil per sample into a heat-resistant container. Samples were loaded into the automated machine, and three reference standards, each consisting of 12% CaCO3 were incorporated at every tenth position. A temperature profile was set to slowly increase from 550-950 °C, which allows carbon-types to be differentiated. Only results from OC, burned at between 550-700 °C are reported here. As above, analysis was performed on 30 pairs of samples, with each sample-pair representing an HB-invaded and nearby reference site.

*Analysis of inequality*

A method for determining inequality between groups of data was initially devised for use in economics studies, to assess disparities between familial incomes (e.g. Lorenz, 1905; Bowman, 1946). Some years later, the initial approach was refined, to enable researchers to determine the extent to which financial inequality existed (Gini, 1912; Ricci, 1916). More recently, however, the technique has been adapted and employed in biological studies, and is now regularly applied in a range of ecological and plant science studies, to investigate specific factors such as differences in size and/or fecundity among groups of plants (e.g. Weiner & Solbrig, 1984; Damgaard & Weiner, 2000), or the efficacy of bacteriological soil amendments to promote plant growth (e.g. Gange & Gadhave, 2018). The approach is exclusive to neither economics or biological disciplines, however, but can be used to evaluate and condense the distribution of any numeric (i.e. interval) dataset into a single value, such as a Coefficient of Variation (CV), or a Gini coefficient. In the latter case, the value ranges from 0 to 1; with the former value indicating no inequality, and the latter indicating total inequality. Despite finding tentative evidence suggesting that sediment flux at HB-invaded areas was higher than reference areas, as indicated by higher standard deviation values for most years (Greenwood *et al*., 2018), the more robust Gini analysis as a measure of inequality represents a means of interrogating datasets such as ours, where extremes of values, signifying exceptionally high-magnitude erosion or deposition, need to be taken into account. This approach has the ability to provide a more robust means of identifying whether inequality in sediment flux exists between HB-invaded and reference areas free of HB invasion. A full description of the calculation of inequality measures and their comparisons is given in Gange & Gadhave (2018). In the present analysis, we used the software packages ‘zar5’ to compare coefficients of variation (Miller, 1991), and ‘ineq’ to calculate Gini coefficients in R 3.4.1, to examine differences in sedimentation inequality.

**Results**

*Sedimentation inequality*

Results from the sedimentation inequality analyses are listed for the Ibach in Table 1 and the variability associated with those datasets are presented as boxplots in Figure 1. Equivalent results from the River Taw are listed in Tables 2 and 3, for winter and summer periods, respectively, and the variability, again, is presented as boxplots in Figure 2A (winter periods) and in Figure 2B (summer periods).

Results from the Ibach (Table 1) show that variability in sediment flux at HB-invaded areas was significantly greater than for reference areas for the years 2012-13, 2013-14 and 2014-15, as indicated by the higher CV values. In contrast, sediment flux at reference areas was significantly greater than HB-invaded areas for the year 2015-16. Although the data set for 2015-16 appears to be more variable in the HB area, this was not significant because of a higher sample size in those areas.

For the River Taw winter data (Table 2 & Figure 2A), variability in sediment flux at HB-invaded areas was significantly greater than reference areas for the year 2013-14 only, as indicated by the higher CV value, whereas for the years 2014-15 and 2015-16, no difference in inequality was found between HB-invaded or reference locations. Despite HB data presented in Figure 2A for the year 2015-16 showing notably higher sedimentation values and apparently much higher variability than reference areas, the difference was not significant between HB or reference areas.

Contrasting results were obtained for the River Taw summer data (Table 3 & Figure 2B), in that variability in sediment flux at HB-invaded areas was significantly greater during the summer period 2015, but there was no difference between either group of transects during summer 2016. As before, sediment flux appeared to be more variable in the HB areas, due to two large outlying data points.

Results from the inequality analyses listed in Tables 1-3 are compared against standard deviation values reported in Greenwood *et al*. (2018) and are listed in Tables 4-6. For brevity, and due to the large number of possible permutations that can arise when comparing outcomes between the two tests, results are depicted in the form of dashes, signifying no difference in sediment flux between HB-invaded or reference sites, and upward and downward arrows, indicating whether sediment flux at HB-invaded or reference transects is higher or lower, respectively. Presented in this manner, results from the two statistical tests on the nine sedimentation datasets can be classified into four distinct groups. For the first group, which consists of three datasets (i.e. Ibach, 2012/13, 2013/14; River Taw, summer 2015), both tests concur that sediment flux at HB-invaded sites was significantly more variable. Secondly, despite three datasets (River Taw, winter periods, 2014/15, 2015/16, and summer 2016) originally returning higher standard deviation values at HB-invaded sites, results from the inequality analyses revealed no significant difference in sedimentation between HB-invaded or reference sites. Thirdly, two datasets (Ibach, 2014/15, and Taw 2013/14) recorded slightly lower standard deviation values at HB-invaded areas. Although these findings were originally interpreted as evidence of more stable sedimentation conditions, results from the inequality analyses contradict those original interpretations, however, and instead reveal that sediment flux was actually significantly more variable at HB-invaded sites. Finally, although a slightly higher standard deviation value was recorded at HB-invaded sites (Ibach, 2015/16) for one dataset, the inequality analysis indicated that sediment flux was significantly more variable at reference sites.

*Physicochemical properties*

Results of inequality analyses for physicochemical soil properties, OC, BD, and P are presented as box plots in Figure 3A, B & C, respectively. Although Greenwood *et al*. (2018) originally reported similar OC content between soils from HB-invaded and reference areas, inequality analysis undertaken in this study indicates significantly higher variability at HB-invaded areas (Figure 3A), with some samples from the HB group recording notably higher OC contents. Although Greenwood et al. (2018) reported significantly higher in-situ BD at reference areas, inequality analysis revealed no difference in variability between HB-invaded or reference areas (Figure 3B). Similar to OC, Greenwood et al. (2018) originally reported no difference in P concentration between soils from HB-invaded or reference areas, but the inequality analysis indicated significantly more variability at HB-invaded areas (Figure 3C), with a number of samples recording very high P concentrations.

**Discussion**

*Standard deviation versus inequality analysis*

Standard deviation tests performed on the two groups of sedimentation data (i.e. HB-invaded and Ref.) composing each of the nine datasets were originally interpreted by Greenwood *et al*. (2018) as evidence of more dynamic sedimentation conditions involving both losses and gains, and ultimately resulting in higher sediment flux. In seven out of the nine datasets, higher standard deviation values reported in Greenwood *et al*. (2018) were recorded at HB-invaded sites, suggesting that sediment flux was considerably greater at those locations than equivalent conditions at nearby reference sites. In contrast, results from inequality analyses presented here found evidence of higher sediment flux in five out of the nine datasets. This more conservative, yet probably more reliable estimate raises the possibility that at least some of the sedimentation datasets are asymmetrically distributed and hence skewed. This, in-turn, raises the question whether standard deviation represents the most appropriate test to measure data-dispersion. Although skewness values were not originally reported in Greenwood *et al*. (2018), results from *post hoc* tests performed on the same sedimentation datasets (data not shown) indicated that five groups returned skewness values in the region of -1.0 or +1.0 or greater (i.e. Ibach: HB, 2012/13, HB and Ref. 2013/14; Taw; HB 2013/14 and Ref., 2015S). In addition, data from two groups of transects recorded values in excess of + 0.6 (i.e. Taw: HB and Ref., 2016S). Given the degree to which some datasets were skewed, therefore, results from inequality analyses support the original interpretation that most areas where HB is well established experience significantly higher sediment flux.

*Physicochemical properties*

Although reasons why OC showed significantly higher variability at HB-invaded sites are not entirely clear, work by others (Andrews *et al*., 2005; Dassonville et al., 2008; Čuda *et al*., 2017) offer potential insights in the form of a number of biochemical mechanisms employed by HB that may explain the presence of high OC content associated with some soil samples from HB sites (Figure 3A). For instance, Andrews *et al*. (2005) reports that not only does OC content account for ca. 40% of dry HB stems, but HB overcomes low irradiance and thrives in deep shade by sequestering high concentrations of potassium (K) and nitrogen (N) from the soil. These nutrients are stored in HB tissue and are used to maintain osmotic pressure as they represent ecologically cheaper alternatives in low light conditions to malate and glucose used by many other plants. Similarly, Dassonville *et al*. (2008) and Čuda *et al*. (2017) report a number of biochemical mechanisms employed by HB plants, that although still poorly understood, are thought to contribute to its invasiveness. These include accumulating the micronutrients, boron (B) which promotes P uptake, and copper (Cu) which promotes photosynthesis and plant metabolism, all of which promote higher net primary productivity (NPP). As HB is an annual plant, those high nutrient concentrations, including OC (Andrews *et al*., 2005), are thought to be rapidly returned to the soil each year through high-litter fall, death and decomposition, the latter of which is facilitated by high concentrations of K and N (Čuda *et al*., 2017). In river catchments such as the Ibach where overbank floods are rare, it is plausible that HB residue decomposes in situ each year, leaving the less labile and more conservative fractions of certain nutrients, such as OC, to enrich the soil. Although clear evidence of this should have been reflected in significantly higher OC content at HB-invaded areas, two reasons may explain why this relationship was absent. Firstly, OC is too heterogeneously distributed and the number of samples collected from HB sites (n=30) was too few to reveal such a trend. Alternatively, net soil loss and the regular exportation of significant amounts of topsoil from HB sites, as reported in two of the four erosion monitoring periods (Greenwood *et al*., 2018), may act as a feedback mechanism that inhibits the accumulation of soil OC stocks.

For in-situ BD, although inequality analysis showed no difference in variability between HB and reference areas (Figure 3B), BD values from HB sites were consistently lower than reference areas and the data-range was notably more compressed. Collectively, these findings are believed to corroborate the significantly lower in situ BD of soils found at HB-invaded areas (Greenwood *et al*., 2018), and reinforces the likelihood that soils where HB are established are consistently less compact. This we attribute to significantly higher sediment flux at certain locations (Tables 1-3), resulting in loose soils that lack both structure and cohesion, and are less able to resist erosion.

Phosphorus values at HB-invaded areas showed significantly higher variability in comparison with reference areas (Figure 3C), with some samples recording concentrations that were almost double the highest concentration at reference areas. By way of explanation, as HB disrupts the relationship between mycorrhizal fungi and perennial plant species, this leaves the fungi, and hence the soil, enriched in P (Tanner & Gange, 2013). Consequently, significantly higher P concentration would be expected in soils from HB-invaded areas. As this was not found (Greenwood *et al*., 2018), reasons why may be similar to the explanations given above for OC. Firstly, the distribution of P may be too heterogeneous, and the number of samples collected from HB sites (n=30) may have been too few to reveal this trend. Secondly, the periodic removal of significant amounts of topsoil from HB sites, as reported in two of the four erosion monitoring periods (Greenwood *et al*., 2018), may act as a feedback mechanism that inhibits the accumulation of P at HB-invaded sites. If this scenario is true, it raises concerns about excessive amounts of nutrient-rich sediment from HB-invaded sites entering fluvial environments (Greenwood *et al*., 2018) and the detrimental effect that this may have on water quality downstream of invaded areas (Arnold & Toran, 2018). Therefore, HB not only significantly alters the microbial and chemical contents of soil in which it grows (Tanner & Gange, 2013, Pattison *et al*., 2016), but it also creates large variability in the latter parameters. This means that any attempt to revegetate sites with native plants following clearance of the weed will, in themselves, be variable in their success (Tanner & Gange, 2013). Much attention has been paid to finding ways to control HB in the UK (Tanner *et al.* 2008), but our results suggest that even if control methods are successful, subsequent colonisation of previously infested sites by natural vegetation may be erratic. Not only does HB reduce beneficial soil microbial populations upon which native plants depend (Pattison *et al*., 2016), the current analysis clearly shows that it creates extreme heterogeneity in soil structure and nutrient content that will lead to highly uneven vegetation re-establishment.

*Colonisation by hydrochory hypothesis*

We believe these additional findings lend support to the ‘colonisation by hydrochory’ hypothesis proposed by Greenwood *et al*. (2018), in which they speculate that localised river-flow conditions concentrate and deposit suspended sediment and plant parts, including HB seeds, at ‘depositional hotspots’ along river margins by hydrochorous processes (Gurnell, 2015). As hydrochory is a predominantly depositional mechanism (Su *et al*., 2019), depressional or slack areas along river margins create calm conditions that act as foci for the enhanced deposition of fine sediment and detritus (Simm & Walling, 1998; Su *et al*., 2019). Due to the adventitious root system of HB plants (Ennos *et al*., 1993) and their tendency to preferentially trap very fine sediment, as reported in Greenwood *et al*. (2018), we propose that this mechanism may initially work in tandem with hydrochory during the early stages of colonisation, resulting in enhanced sedimentation but creating a predominantly stable environment. In times of fluctuating river-flow, however, the possibility exists that areas where HB is well established may periodically experience high-magnitude erosion, or higher-than-background sediment flux, due to factors such as localised topography, and the fact that regular inputs of fresh sediment deposits remain loose and more susceptible to erosion, particularly when understory vegetation is absent or depleted. This is supported by the significantly lower in-situ BD at HB-invaded areas (Greenwood *et al*., 2018), and the constrained range of the BD dataset from HB-invaded sites (Figure 3B). Variable sedimentation conditions such as those described above are therefore believed to offer a plausible reason as to why erosion from HB-invaded areas was not significantly greater than erosion from reference areas for all years when monitoring was undertaken and may explain the inconsistent results reported by Greenwood *et al*. (2018).

**Conclusion**

Although the dynamics of river bank erosion means that results are difficult to generalise, the findings thus far lead us to conclude that hydrochory probably deposits HB seeds in low-flow or retentive areas along river margins, and this dictates where HB initially becomes established. Conflicting erosion results from the seven-year monitoring record suggest that, during the early colonisation phase, sedimentation processes remain generally comparable with areas free of HB. However, as the number/density of HB plants increases, perennial vegetation is increasingly displaced until a tipping-point is reached, whereby localised factors such as river-flow conditions and topography conspire to determine whether the net sedimentation process over any given winter period is one of losses or gains. Against this possibility, many authors have used the term ‘ecosystem engineer’ to describe the generally positive role that most native riparian vegetation plays in stabilising river systems after disturbance, by attenuating river-flow, encouraging sediment deposition, and contributing to the creation of different habitat-types (Gurnell, 2015; Polvi & Sarneel, 2017). As an alien species, however, although HB plays no role in determining where initial colonisation occurs, its invasive tendencies and ability to displace perennial vegetation (Tickner *et al*., 2001) led Greenwood & Kuhn (2015) to suggest that after seasonal die-off, the prolonged absence of HB, coupled with suppressed perennial vegetation cover elevates the risk that many invaded areas may experience significant soil loss and/or higher-than-background sediment flux in comparison with uninvaded areas during some, but not all winter periods. In doing so, we conclude that HB could arguably represent the antithesis of a river system engineer.

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**Author contributions**

PG and NJK contributed to the design of the erosion measurement experiments, which were executed by PG. ACG conducted data analyses, and PG wrote the paper, with substantial contributions from ACG and NJK.

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**Tables**

Table 1. Results of sedimentation inequality analyses between HB-invaded and reference sites for the Ibach river

|  |
| --- |
| **River Ibach - Winter Data** |
| **Year** | **P. Value** | **Coefficient of Variation (CV)** | **Inequality** | **Gini Index** | **Inequality Direction** | **N** |
| **HB** | **Ref.** | **HB** | **Ref.** |
| 2012/13 | <0.05 | 0.31 | 0.22 | Sig. | 0.16 | 0.12 | HB > Ref. | 840 |
| 2013/14 | <0.05 | 0.36 | 0.16 | Sig. | 0.19 | 0.09 | HB > Ref. | 600 |
| 2014/15 | <0.05 | 0.23 | 0.06 | Sig. | 0.13 | 0.03 | HB > Ref. | 500 |
| 2015/16 | <0.05 | 0.13 | 0.19 | Sig. | 0.07 | 0.11 | HB < Ref. | 1740 |

Table 2. Results of sedimentation inequality analyses between HB-invaded and reference sites for the River Taw, winter periods

|  |
| --- |
| **River Taw - Winter Data** |
| **Year** | **P. Value** | **Coefficient of Variation (CV)** | **Inequality** | **Gini Index** | **Inequality Direction** | **N** |
| **HB** | **Ref.** | **HB** | **Ref.** |
| 2013/14 | <0.05 | 0.49 | 0.18 | Sig. | 0.25 | 0.11 | HB > Ref. | 680 |
| 2014/15 | >0.05 | 0.38 | 0.30 | Not Sig. | 0.21 | 0.17 | HB = Ref. | 840 |
| 2015/16 | >0.05 | 0.39 | 0.45 | Not Sig. | 0.22 | 0.22 | HB = Ref. | 540 |

Table 3. Results of sedimentation inequality analyses between HB-invaded and reference sites for the River Taw, summer periods

|  |
| --- |
| **River Taw - Summer Data** |
| **Year** | **P. Value** | **Coefficient of Variation (CV)** | **Inequality** | **Gini Index** | **Inequality Direction** | **N** |
| **HB** | **Ref.** | **HB** | **Ref.** |
| 2015 | <0.05 | 0.42 | 0.18 | Sig. | 0.24 | 0.09 | HB > Ref. | 560 |
| 2016 | >0.05 | 0.39 | 0.28 | Not Sig. | 0.21 | 0.14 | HB = Ref. | 440 |

Table 4. Standard deviation values compared against results of sedimentation inequality analyses for the Ibach river, and results of both tests summarised

|  |
| --- |
| **Ibach** |
| Year | St. Dev. | Gini Analysis | **Summary** |
| HB-invaded | Ref. | St. Dev. | Inequality |
| 2012/13 | 6.00 | 3.02 | Sig. higher sed. flux: HB-invaded | ↑HB | ↑HB |
| 2013/14 | 8.23 | 3.06 | Sig. higher sed. flux: HB-invaded | ↑HB | ↑HB |
| 2014/15 | 4.76 | 5.39 | Sig. higher sed. flux: HB-invaded | ↓HB | ↑HB |
| 2015/16 | 1.93 | 1.29 | Sig. higher sed. flux: Ref | ↑HB | ↑ref |
| ↑ = higher; ↓ = lower; ─ = no difference |  |  |  |

Table 5. Standard deviation values compared against results of sedimentation inequality analyses for the River Taw site (winter data) and results of both tests summarised

|  |
| --- |
| **River Taw Winter** |
| Year | St. Dev. | Gini Analysis | **Summary** |
| HB-invaded | Ref. | St. Dev. | Inequality |
| 2013/14 | 33.62 | 33.66 | Sig. higher sed. Flux: HB-invaded | ↓HB | ↑HB |
| 2014/15 | 31.41 | 20.67 | HB = Ref. | ↑HB | ─ |
| 2015/16 | 47.06 | 11.93 | HB = Ref. | ↑HB | ─ |
| ↑ = higher; ↓ = lower; ─ = no difference |  |  |  |

Table 6. Standard deviation values compared against results of sedimentation inequality analyses for the River Taw site (summer data) and results of both tests summarised

|  |
| --- |
| **River Taw Summer** |
| Year | St. Dev. | Gini Analysis | **Summary** |
| HB-invaded | Ref. | St. Dev. | Inequality |
| 2015 | 21.95 | 10.46 | Sig. higher sed. Flux: HB-invaded | ↑HB | ↑HB |
| 2016 | 31.71 | 6.32 | HB = Ref. | ↑HB | ─ |
| ↑ = higher; ↓ = lower; ─ = no difference |  |   |

**Figure Captions**

**Fig. 1** Box plots showing the range in sedimentation flux for areas invaded by Himalayan balsam (HB) and uninvaded areas (Ref) for the River Ibach, Switzerland across four years. The horizontal line within the box is the median, while edges of the box represent the inter-quartile ranges. The whiskers depict 1.5 x the inter-quartile ranges, while points depict outliers beyond the whiskers.

**Fig. 2** Box plots showing the range in sedimentation flux for areas invaded by Himalayan balsam (HB) and uninvaded areas (Ref) for the River Taw, UK, across three winter seasons (A) and two summer seasons (B). The horizontal line within the box is the median, while edges of the box represent the inter-quartile ranges. The whiskers depict 1.5 x the inter-quartile ranges, while points depict outliers beyond the whiskers.

**Fig. 3** Boxplots showing variability in soil carbon (A), bulk density (B) and phosphorus (C) in areas invaded by Himalayan balsam (HB) and uninvaded (Ref.) areas at the River Ibach, Switzerland.

Figure 1

Figure 2



Figure 3

