Resources, Environment and Sustainability 7 (2022) 100050

Contents lists available at ScienceDirect

Resources, Environment and Sustainability

journal homepage: www.elsevier.com/locate/resenv

Resource recovery and freshwater ecosystem restoration — Prospecting for phytoremediation potential in wild macrophyte stands

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ARTICLE INFO

Keywords:
Phytoextraction
Eutrophication
Diffuse pollution
Nature-based solutions
Circular economy

ABSTRACT

The primary aim of this study was to understand the factors, e.g., harvest frequency and plant community type, that can facilitate optimising phytoextraction in wild macrophyte communities as part of a strategy for water-quality improvement and resource recovery. This was achieved by surveying wild macrophyte communities and quantifying standing stocks of key nutrient pollutants such as N and P, and a range of other recoverable macro and micro-nutrients (Ca, K, Mg, Cu, Fe, Mn, Mo, Zn, Na and Cr). By scaling-up the pollutant export potential over a decade, it was determined which harvest strategy and plant community types provide the greatest levels of nutrient export. Grime's CSR plant strategy framework was used to categorise each surveyed community, where large-statured, higher biomass producing competitor and stress tolerator-type communities were compared with ruderal-type communities that have rapid growth and high nutrient acquisition but smaller standing biomass and statures. High biomass plant communities containing competitor or stress tolerator species, produce greater standing stocks of macronutrients (such as N and P) for harvesting, while yields of micronutrient-type pollutants are more likely to be influenced by specific physiological traits that determine leaf tissue concentration. Utilising a high frequency harvest regime over a multi-year time scale suggested that small fast-growing ruderals could yield 4–6 times the concentration of macronutrients and micronutrients for export compared to competitor or stress tolerator-dominated communities e.g., P yields from ruderals were 25 g/m^2 versus 5 g/m^2 from competitor/stress tolerator-dominated communities. These results emphasise the need to consider both the plant community and the harvesting regime when using phytoextraction as a management tool. We anticipate that these results will help guide environmental managers in their approach in developing circular economy schemes that improve water quality through nutrient export.

1. Introduction

Freshwaters are under increasing pressure from diffuse pollutants such as phosphorus (P), nitrogen (N) and potentially toxic metals (Berger et al., 2017). These pollutants can impair ecosystem health, decrease water quality for domestic and agricultural use, and reduce access to water for recreation (Berger et al., 2017; Posthuma et al., 2019). Following rainfall, agricultural and semi-rural landscapes often generate run-off contaminated with synthetic fertilisers, soil, sediment, manure, and pesticides (Foley et al., 2005). Vegetated buffer zones and nutrient management planning (including appropriately-timed fertiliser application and loading) are best management practices that are critically important for reducing contaminant transfer and delivery to freshwaters (Lam et al., 2011). However, if mitigation measures become compromised and/or there are stores of legacy pollutants in the catchment, remedial systems to remove pollutants are required (Jarvie et al., 2013).

Aquatic phytoremediation is a promising nature-based solution (NBS) that capitalises on the ability of macrophytes to take-up and sequester pollutants from water and sediments and thus improve water quality (Newete and Byrne, 2016). The primary foci of aquatic phytoremediation as an application has been the construction of artificial wetlands, i.e., the deliberate planting, and deployment of macrophytes in shallow freshwaters (Fletcher et al., 2020). An alternative approach is to harvest existing stands of wild macrophytes in locations where diffuse, point source, and legacy pollution is a problem. Here, the aim is to harvest and remove pollutants sequestered in above-ground plant tissue, therefore reducing the return of nutrients back to the water following plant die-back (Zhou et al., 2017). Pollutant sequestration by macrophytes from sediment and/or the water column is recognised as an important pathway for pollutant removal from freshwater (Preiner et al., 2020; Zhang et al., 2007), e.g., harvesting macrophyte biomass can offset incoming nutrient inputs into water bodies by over 50% (Bartodziej et al., 2017; Huser et al., 2016). Furthermore, through systematic harvesting and removal of macrophyte biomass there are opportunities to rebalance nutrient losses from

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https://doi.org/10.1016/j.resenv.2022.100050
Received 3 September 2021; Received in revised form 9 February 2022; Accepted 13 February 2022
Available online 22 February 2022

terrestrial ecosystems as part of a circular agronomic model (Quilliam et al., 2015). This represents a crucial opportunity to reduce demands on raw materials given that many of these pollutants are derived from compounds manufactured in an energy and resource intensive way (Jones et al., 2013), or their supply is finite. Phytoextraction is the specific phytoremediation process whereby targeted pollutants are taken-up and assimilated into macrophyte tissue, which is then harvested to enable the pollutants to be disposed of or targeted for resource recovery.

Our current understanding of phytoextraction potential of macrophytes comes from quantifying tissue concentrations from either controlled experiments or surveys where specific species or locations were targeted (Petrů and Vymazal, 2018; Kuiper et al., 2017; Vinten and Bowden-Smith, 2020). While these approaches provide important information for specific phytoremediation candidates, they do not necessarily enable larger-scale harvesting strategies to be explored at multiple locations in a catchment or landscape where water body characteristics, pollutant types and sources, and vegetation will naturally vary. Furthermore, focusing on tissue concentrations alone to measure phytoextraction can give a false impression of the quantity of pollutant that can be removed via plant harvesting as it does not account for the amount of biomass available (Vymazal, 2016). Standing stock measures of pollutants give a more representative quantity of the total phytoextraction potential from aquatic systems. However, this can also lead to the misunderstanding that species with larger biomass should always be the *de facto* choice when selecting plants for phytoextraction. When harvesting is employed as a long-term strategy, cross-cutting factors such as pollutant type, harvest strategy and dominant plant growth strategies can impact the total yield of target chemicals over several years. Disentangling the importance of biomass and tissue concentration when determining standing stocks for different pollutants is an important consideration for pollutant removal and resource recovery. Plant life strategies within the CSR plant growth framework proposed by Grime (Pierce et al., 2012; Grime, 1977) are closely related to nutrient recovery from harvested plants because of contrasting standing biomasses and nutrient acquisition between competitor, stress-tolerator, and ruderal dominated communities (Grime, 1977; Pierce et al., 2017). Therefore, classifying plant communities using this approach helps to generalise results across different communities and environments. It has been recognised that macrophytes are generally not well represented by strongly stress-tolerator selected species (i.e., with conservative resources economics) (Pierce et al., 2012). In this study, most of the studied communities were predominantly competitor dominant with some species that exhibited variation towards the stress tolerance strategy which ultimately meant communities were grouped by competitor and stress tolerators together (C/S), versus ruderals (R).

The primary aim of this work was to understand the factors, e.g., harvest frequency and plant community type, that can help optimise phytoextraction in wild macrophyte communities as part of a strategy for water quality improvement and resource recovery. This was achieved by surveying wild macrophyte communities and quantifying standing stocks of key pollutants. Scaling-up the pollutant/nutrient export potential over a decade can enable an estimate of which plant community types can provide the greatest levels of nutrient export depending on the chosen harvest strategy. Investigating the relationship between macrophyte tissue concentrations of pollutants and biomass will increase our understanding of how these two factors influence the standing stock concentrations of a suite of pollutants and facilitate the targeting of natural vegetation for harvesting.

2. Methodology

2.1. Survey sample strategy

A field survey was designed to capture a wide range of freshwater sites (including both lotic and lentic systems) and characterise macrophyte phytoextraction potential within an area dominated by lowland agriculture and urban land uses. In total, 21 individual sites were selected using aerial photography and Ordnance Survey maps to identify areas of macrophyte coverage over a range of land uses (Fig. 1; Table S1). Sampling was undertaken between July and September 2017, to coincide with peak biomass. A stratified random sample strategy was used where visually homogenous stands of macrophytes greater than 1 m^2 were targeted (the average stand size was 150 m^2). To avoid sampling terrestrial plants, only macrophytes growing in at least 3 cm of standing water were sampled.

A 0.25 m^2 quadrat was placed within each stand and all macrophytes were identified, the total coverage recorded, and the Domin scale used to estimate community composition. Simulating a realistic harvesting regime, only the more accessible above ground plant parts were removed from within each quadrat for biomass and tissue concentration analyses. This included stems and leaves for emergent and submerged plants, whilst for free-floating plants that lacked substrate anchorage the whole plant was harvested. The total area of the stand was estimated, either on site or, for larger stands, using aerial photographs (Google Earth Pro 3.0). Harvested plant tissue was washed with tap water to remove sediment and oven dried at 75 °C until a constant dry weight was achieved.

At each sample quadrat location, a 60 ml water sample was taken inside the quadrat at a depth of approximately 5 cm for subsequent water quality analyses including a range of parameters detailed in Section 2.2. A second 60 ml water sample was taken either upstream or in an open water location to compare water quality in an area not directly influenced by the macrophyte stand. All water samples were vacuum filtered through 1 μm pore-size Whatman glass microfiber filters (Whatman PLC, Buckinghamshire, UK), to maximise representation of dissolved element fractions, within 4 h of collection to remove particulate material. Filtered samples were then preserved for bulk analysis by freezing at −20 °C. Unfiltered samples were used to determine turbidity using a LP2000 turbidity meter (Hanna Instruments, Bedfordshire, UK). Conductivity, pH and water temperature were quantified in the field using a Hanna HI 98129 combi-meter (Hanna Instruments, Bedfordshire, UK).

2.2. Determination of pollutant concentrations in water and plant tissue samples

A SEAL Analytical AA3 Continuous Segmented Flow Autoanalyzer was used for determination of nitrogen species in the water samples (NH_3, NO_2^-, NO_3^-) using SEAL analytical method No. G-171-96 Revision 8 and No. G-172-96 (Revision 9). Total P (< 1 μm particle size) and metalloid elements in water were quantified by inductively coupled plasma spectrophotometry (ICP-Optical Emission Spectrometer, Thermo Scientific iCAP 6000 Series ICP; Thermo Scientific, UK). Dried plant tissue was weighed to quantify total biomass per quadrat; from this, approximately 10% by weight of each sample was selected and pulverised using a RETSCH RS200 vibratory disk mill (RETSCH, Germany) to obtain material for tissue concentration analysis. Milled subsamples were either analysed for total C and N using a C:N analyser (FlashSmart NC ORG, ThermoFisher Scientific, UK), or microwave-digested with 70% nitric acid and analysed for P and metalloid element concentration using ICP spectrophotometry. Calcium (Ca), potassium (K), magnesium (Mg), nitrogen (N), and phosphorus (P) are collectively referred to here as ‘macronutrient-type pollutants’ while micronutrients including copper (Cu), iron (Fe), manganese (Mn), molybdenum (Mo), zinc (Zn), sodium (Na) and chromium (Cr) are collectively referred to as ‘micronutrient-type pollutants’.

2.3. Community ecology variables and nutrient standing stocks

Plant communities were categorised by the dominant plant growth strategy following Grime’s C (competitor), S (stress tolerator) and R (ruderal) (CSR) plant growth strategy framework (Pierce et al., 2017).

Fig. 1. Location of sample sites across central Scotland, numbers correspond to site information in Table S1. The map extents North 55°58’13.7”–56°11’14.1”, and West 04°02’20.3”–03° 20’ 58.1”.

The component plants from each community were assigned to their primary growth strategies (i.e. C, S and R) on a continuous scale (Pierce et al., 2017). Of the 30 species recorded, only four could not be matched directly with those documented by Pierce et al. (2017); therefore, these species were matched to a closely related species with a similar growth form, habitat, and life history. For each quadrat, the proportion of the primary growth CSR strategies within the community, weighted by the Domin cover scores of the component species, were calculated following the approach of Willby et al. (2001).

Once the proportion of C, S or R strategies in each community were known, each community quadrat sample was assigned a category for the purpose of analysis: competitor/stress tolerator (C/S) or ruderal (R). These were chosen as categories because these life strategies relate strongly to harvest potential. Ruderals are adapted to disturbance and are fast growing which means there is the potential for multiple harvests per year, whereas for C/S species any regeneration after loss of biomass is likely to be much slower, e.g., taking several seasons growth to reach a similar biomass. Where the proportion of ruderals was greater than 50% these samples were assigned to the ruderal category. Where there were similar proportions of each strategy a decision was made on what plant species would determine the harvest regime. For example, if abundance of tall macrophytes (e.g., *Glyceria maxima*) was greater than 30% then these were assigned to the C/S, as the harvesting strategy would normally be defined by the greater presence of larger statured plants because of their higher contribution of biomass.

The total standing stock of each pollutant per 0.25 m^2 quadrat was calculated by multiplying the community nutrient tissue concentration (mg/g) by the total community biomass (g). Standing stocks were multiplied by 4 to present results on a g/m^2 basis so they could easily compared with existing literature. Total standing stock is used as the main indicator of phytoextraction potential as it incorporates the realised measure of pollutant export potential (Vymazal, 2016). To assess the interaction between harvest frequency and pollutant removal/nutrient export the standing stocks from the survey were multiplied-up over a 10-year time scale with three harvesting scenarios considered: (1) High frequency (HF) in which C/S communities are harvested every three years (to allow recovery time), and ruderal communities harvested three times annually; (2) Low frequency (LF) in which C/S communities are harvested every three years and ruderal communities are harvested only once annually; and (3) Medium frequency (MF) with C/S communities harvested every year but with a 40% decline in productivity applied to all values after the first harvest, and ruderal communities harvested once annually to simulate a scenario where one annual harvesting is carried out for all community types (Table 1). These scenarios have been informed by existing studies of plant re-growth following harvesting and the number of harvests, and productivity decline of C/S communities in the MF is a realistic parameter for the harvest strategies (Nassi o Di Nasso et al., 2010; Niemiec et al., 2019; Jakubowski et al., 2010; Avellán and Gremillion, 2019). Apart from the C/S communities in the MF scenario, a constant standing stock return is assumed in each post-harvest period to fit with observations of the effects of plant harvesting or other management activities (e.g., ditch dredging and weed-cutting).

2.4. Data analysis

All data analysis was carried out in R studio version 4.0.3 (R Core Team, 2019). For group comparisons, non-parametric Wilcoxon tests were performed as data did not conform to the assumptions required for parametric tests. Similarly, Spearman rank-order correlation coefficients were calculated to assess relationship between biomass, tissue concentrations and standing stocks.

To assess communities that may have hyperaccumulation potential the bioconcentration factor (BCF) was calculated for each community using the following equation:

$$BCF = \frac{Pollutant\ concentration\ in\ plant}{Pollutant\ concentration\ in\ water}$$

The BCF is most applicable to understanding the accumulation of metalloid elements in plant tissue, therefore has only been applied to the micronutrient-type pollutants. The ratio is indicative of phytoextraction potential; critical values of the BCF > 25 indicate good accumulation, whilst a BCF > 50 shows potential for hyperaccumulation (van der Ent et al., 2013). Mean standing stocks of the most common macrophytes were also calculated following the approach detailed in Section 1.0 of the supplementary information to understand which species within each community may be useful for resource recovery.

Table 1
Harvest scenarios including frequency of harvest, total harvests per scenario and reduction in productivity for ruderals (R) and competitor/stress tolerator (C/S) community types.

Harvest scenario	Frequency of harvest		Total harvest per scenario		Reduction in C/S productivity
	Ruderals	C/S	Ruderals	C/S	
High frequency (HF)	3 harvests per year	1 harvest every 3 years	30	4	–
Medium frequency (MF)	1 harvest per year	1 harvest per year	10	10	40%
Low frequency (LF)	1 harvest per year	1 harvest every 3 years	10	4	–

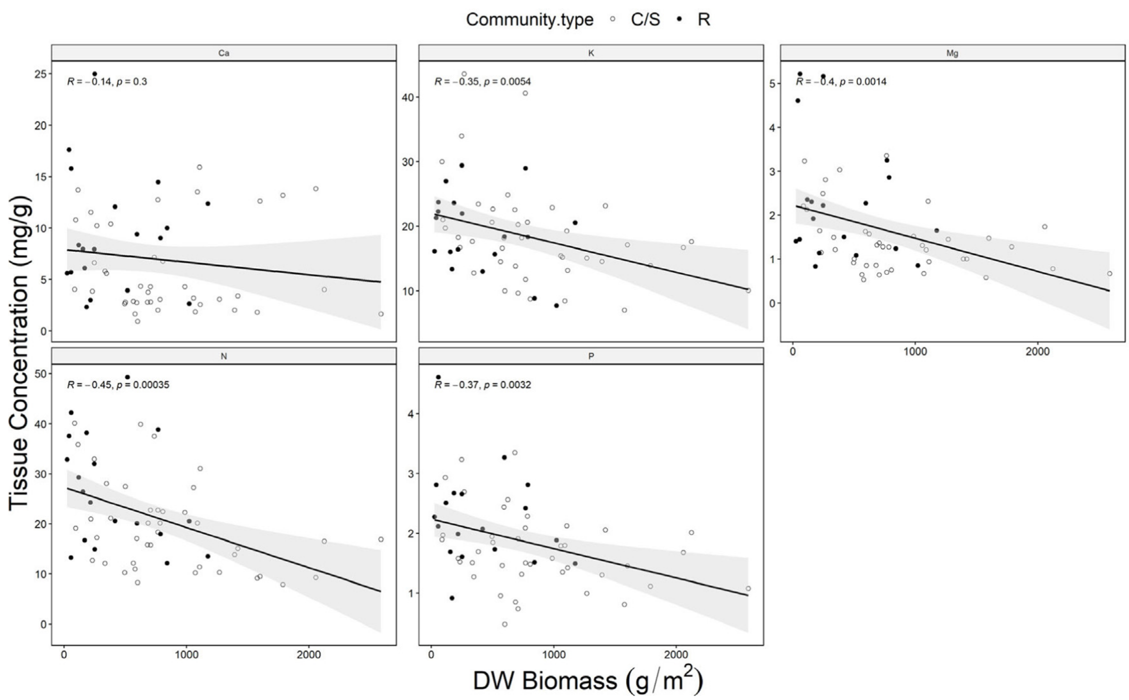

Fig. 2. Relationship between dry weight biomass (g/m^2) and tissue concentrations of macronutrients (mg/g) (n = 61). Datapoints represent competitor/stress tolerator (C/S) communities (open circles) and ruderals (R) communities (filled circles). Regression line and 95% confidence intervals is calculated for the whole dataset including both community types.

3. Results

3.1. Co-variation between biomass, pollutant tissue concentrations and standing stocks

Tissue concentrations of most macronutrient-type pollutants had significant weak, to moderately strong, negative correlations with community biomass (R^2 between 0.28–0.45, P < 0.05) (Fig. 2). The concentration Ca had both a weak and non-significant correlation with community biomass (P > 0.05). C/S communities had a significantly higher community biomass compared to ruderal communities, with a greater value range (P < 0.05) (Fig. 2; Fig. 3; Figure S2). Ruderal communities generally had significantly higher tissue concentrations than C/S communities (Figure S3 & S4). Specifically, Ca, Mg, N, P, Fe and Mn tissue concentrations were significantly greater in ruderal communities compared to competitor/stress tolerators (P < 0.05) (Figures S3 & S4). Overall, communities with a lower biomass tended to have higher tissue concentrations (Fig. 2; Fig. 3). However, for each pollutant, tissue concentration did not significantly correlate with community biomass (P > 0.05) (Fig. 3). Broadly, there was little difference in magnitude between the two community types in micronutrient-type pollutant tissue concentrations (Figure S4).

Standing stocks of macronutrient-type pollutants had strong, and positive, significant correlations with biomass (Fig. 4). The standing stocks of K, N and P in C/S communities were significantly higher than in ruderal communities (P < 0.05) (Figure S5), although there were no differences between the two community types for standing stocks of Ca and Mg. Standing stocks of Cr, Cu and Mo micronutrient-type pollutants were positively correlated (R^2 = ~0.5) with biomass (Fig. 5) (P < 0.05), while Fe, Mn, and Zn standing stocks were not correlated with biomass (P < 0.05). However, Na was an exception, with a weak negative correlation with biomass (Fig. 5). Furthermore, except for Cr, there were no significant differences in median pollutant standing stocks between C/S tolerator communities and ruderal communities (P < 0.05) (Figure S6). There were several communities that had a relatively low to medium biomass but still had higher pollutant standing stocks than communities with the highest biomass, e.g., communities with very low biomass also had higher Na, Cu, Mo and Zn standing stocks (Figs. 4 & 5). Where the correlation between biomass and tissue concentration was weak, predicting standing stocks of macro or micronutrients was more difficult. In these cases, it was possible to highlight those communities with phytoremediation potential using the bioconcentration factor in Section 3.2.

3.2. Bioconcentration factors and common phytoextractor species

For micronutrient-type pollutants, the application of the bioconcentration factor (BCF) suggests that several plant communities could be acting as hyperaccumulators (Table 2). The BCFs of these communities exceeded the critical thresholds as stipulated in Table S3 and contained high tissue concentrations of Fe, Cu and Mn compared to the relative water concentrations (Table 2; Table S3). Importantly, none of these communities contained more than two different macrophyte species; however, overall species richness per community did not have an influence on tissue concentration or nutrient standing stocks (Figure S7). Using the most common species found in the survey to generalise,

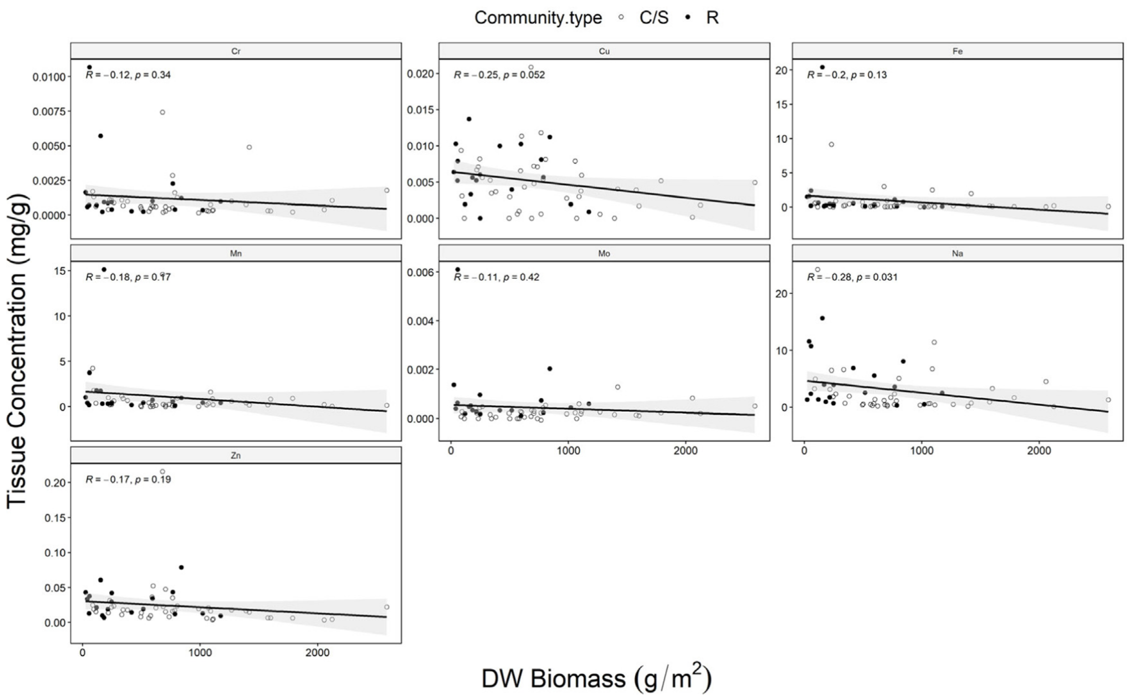

Fig. 3. Relationship between dry weight biomass (g/m^2) and tissue concentrations of micronutrient-type pollutants (mg/g) (n = 61). Datapoints represent competitor/stress tolerator (C/S) communities (open circles) and ruderals (R) communities (filled circles). Regression line and 95% confidence intervals was calculated for the whole dataset including both community types.

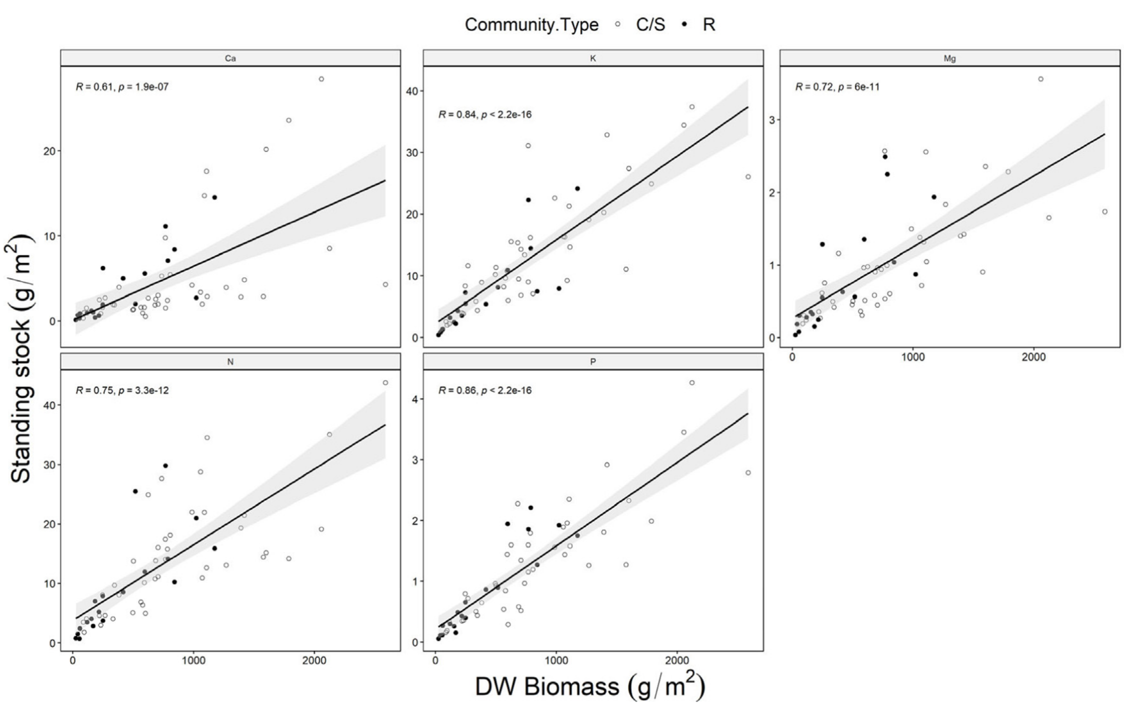

Fig. 4. Relationship between dry weight biomass (g/m^2) and standing stocks (g/m^2) of macronutrient-type pollutants (mg/g) (n = 61). Datapoints represent competitor/stress tolerator (C/S) communities (open circles) and ruderals (R) communities (filled circles). Regression line and 95% confidence intervals is calculated for the whole dataset including both community types.

specific phytoextractor species indicated that *Phragmites australis* and *Glyceria maxima* ($P < 0.05$) had the greatest standing stocks of P and N ($P > 0.05$) (Figure S8). For the micronutrient-type pollutants there was insufficient evidence to determine if any of the species were optimal phytoextractors across the environments studied (Figure S9), e.g., although mean standing stocks were generally higher for *P. australis*, *G. maxima* and *T. latifolia* this was not statistically significant.

3.3. Potential pollutant yields over 10-years by harvest scenario

In the HF scenario, ruderal dominated communities had significantly higher yields of both N and P compared to competitor/stress tolerator communities over the 10-year timescale, ($P < 0.05$) (Fig. 6). Differences between the two community types were most pronounced for this harvest strategy, with ruderal communities providing a mean

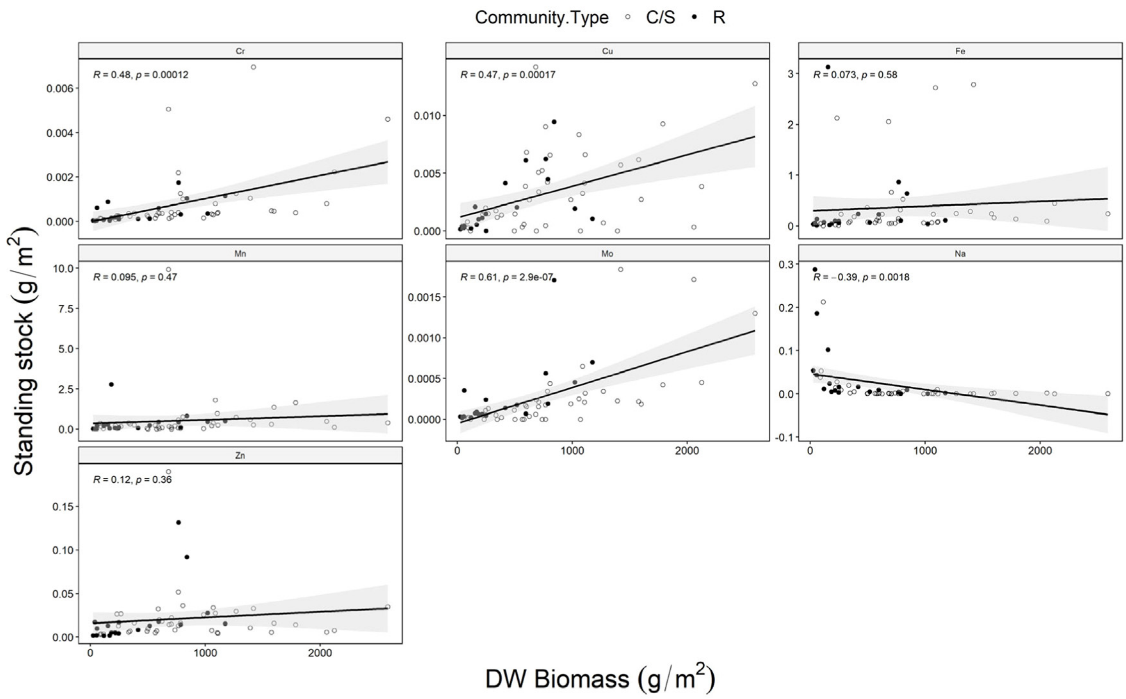

Fig. 5. Scatterplots showing relationship between dry weight biomass (g/m^2) and standing stocks (g/m^2) of pollutants) ($n = 61$). Datapoints represent competitor/stress tolerator (C/S) communities (open circles) and ruderals (R) communities (filled circles). Regression line and 95% confidence intervals is calculated for the whole dataset including both community types.

Table 2
Community and species associated with high bioconcentration factors (full table of BCFs in Table S3).

Element	Community
Fe	*Glyceria maxima* *Azolla filiculoides* *Callitriche stagnalis* *Phalaris arundinacea* and *Glyceria maxima* (mixed stand)
Cu	*Azolla filiculoides* *Typha latifolia* and *Lemna minor* (mixed stand) *Alisma plantago-aquatica* and *Equisetum fluviatile* (mixed stand) *Juncus effusus*
Mn	*Glyceria maxima* *Azolla filiculoides* *Potamogeton natans* and *Alisma plantago-aquatica* (mixed stand)

yield of N and P that was four times greater than competitor/stress tolerator communities. Conversely, for both the MF and the LF harvest strategies there was no significant difference in median total yield of N and P between the two community types. Across each harvest scenario the patterns described above were similar for all studied pollutants including Ca, Cr, Cu, Fe, K, Mg, Mo, and Zn (Figure S10). The magnitude of the difference between the two communities varied, with ruderal communities having 2–6.5 times greater mean yields in the HF scenario than the competitor/stress tolerator communities. Mn was an exception to this as there were no significant differences between the two community types in mean yield across all harvest strategies. Median Na yields were significantly higher in each harvest scenario for ruderal communities ($P < 0.05$) (Figure S10), with the HF strategy yielding the most Na over 10 years compared to all the other scenarios.

4. Discussion

Over a simulated 10-year period, higher frequency harvesting of plant communities comprised primarily of ruderal species provides an opportunity for macro and micronutrient pollutant removal, and the potential for subsequent resource recovery. Harvest frequency is thus a crucial confounding factor that can alter long-term returns from phytoextraction, although it does depend on the plant growth strategy. Reduction in the number of simulated harvests of ruderal-dominated communities in the MF and LF scenarios led to the biomass-driven standing stocks of C/S communities becoming more important for determining returns, therefore there were no significant differences with ruderals in the other scenarios of harvest frequency.

To determine standing stocks (and therefore resource recovery potential) total biomass has more importance for macronutrient-type pollutants, as opposed to micronutrient-type pollutant stocks, suggesting that biomass accumulation is a critical trait for the export and recovery of N, P, K, Mg, Ca and indeed Mo (a micronutrient) (Zhou et al., 2017). This relates to the basic trade-off in plant growth strategies between energy invested in reproduction versus vegetative growth to monopolise light (Craft, 2016). Competitor-dominated communities in this study, such as those including *Typha latifolia*, *Phragmites australis* and *Glyceria maxima* generally have large statures and are more likely to have larger biomass, and therefore higher standing stocks of macronutrients. Conversely, the ruderal plant communities generally have smaller statures leading to a reduced macronutrient standing stocks. Higher tissue concentrations, driven by physiological traits, phenotypic plasticity and concentrations of pollutants in the growth media are likely to be more important in determining standing stocks of micronutrient-type pollutants Cu, Fe and Zn (Padmavathiamma and Li, 2007; Ali et al., 2013). If harvesting regime over time is not considered, macronutrient recovery is more easily achieved simply by harvesting high biomass plants, while for micronutrients closer attention to species-specificity is required, particularly for identifying hyperaccumulators.

By optimising a HF harvest strategy, ruderal communities have the potential to return substantial yields compared to harvesting C/S communities. Wetland ruderal plants are characterised by fast growth rates and leaf turnover, and rapidly proliferating root systems with a high capacity for the absorption of limiting nutrients (Willby et al., 2001). Consequently, they have higher tissue concentrations and can

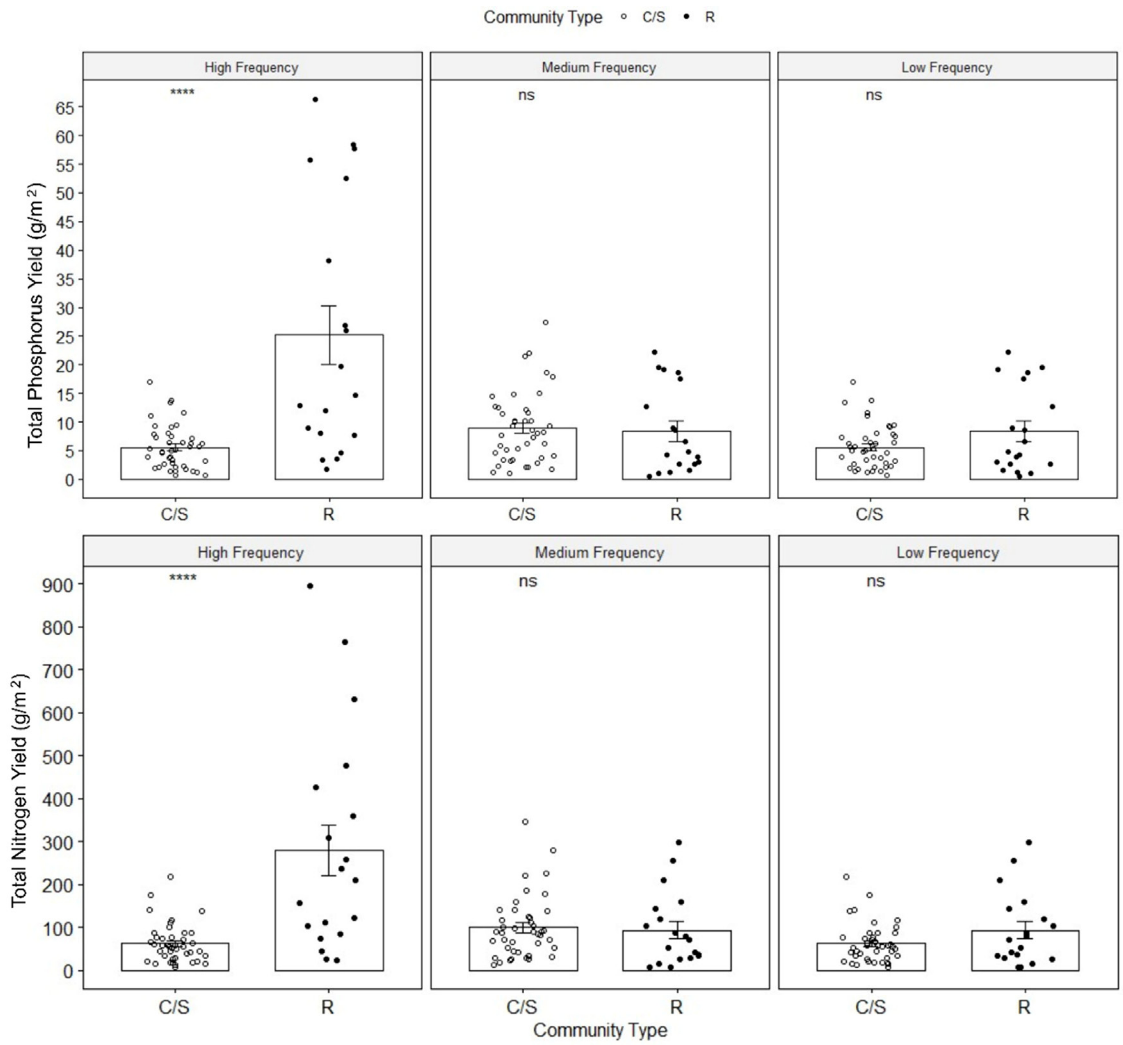

Fig. 6. Scenario-based differences over ten years in total yield of N and P between competitor/stress tolerators (C/S) (n = 42) and ruderal (R) (n = 19) communities. Error bars show the SE of the mean. $P \leq 0.0001$ (****) and 'ns' indicates no significant difference between the two community types.

be harvested multiple times annually; thus, previous studies utilising bi-weekly harvests have strongly advocated floating aquatic ruderals such as *Azolla* as ideal phytoextractors (Tang et al., 2017). While extra harvests could theoretically increase nutrient acquisition, an intense harvest regime with a lower recovery time may result in lower biomass gains over the full season and reduce overall nutrient gains (Bal et al., 2017). An early harvest at the start of the growth season can enable biomass to recover in 3–6 weeks and facilitate plant recovery time to enable further harvests (Bal et al., 2006; Cooke et al., 1990), and in tropical regions with reduced seasonality there is great potential for multiple annual harvests (Vymazal, 2007).

Less intensive harvest regimes such as the MF and LF scenarios that do not capitalise on the rate of ruderal re-growth reduce the benefit of targeting macrophyte communities, or deliberately planting specific communities for pollutant removal and resource recovery. However, the cost–benefit of harvesting multiple times annually is an important consideration and depends on the method of harvest. The approach taken here to harvest only the accessible plant parts means the results are applicable to conventional mechanical harvesting, one of the most economical methods available (Quilliam et al., 2015). C/S communities are less adapted to continual disturbance due to slower relative rates of biomass acquisition and the requirement to aerate a large below-ground biomass. Hence, returns from these more slow-growing plants would likely be diminished by employing an annual harvest (Atkinson et al., 2014). Therefore, wherever high frequency harvests are not planned, other priorities should guide the choice of plant communities for harvesting to generate multiple benefits. Considerations might include the method of harvesting and what other collateral benefits or ecosystem services can be gained. For example, nesting habitat could be provided for waterfowl by planting and periodically harvesting large statured C/S communities containing *Glyceria maxima*.

There are potential negative consequences to harvesting freshwater macrophytes such as the resuspension of nutrients, loss of structural habitat, increased bank erosion, and the risk of shifting lacustrine systems from clear-water to phytoplankton-dominated (Sayer et al., 2010; Habib and Ar, 2016; Soana et al., 2018). However, with an appropriate site characterisation and sustainable harvest regime many negative effects can be mitigated (Kuiper et al., 2017; Kohzu et al., 2019). Broadly speaking, the results presented here demonstrate the significant potential for water quality improvements and resource recovery by harvesting macrophytes from aquatic systems. This concept has shown particular success when coupled with forms of sustainable agriculture less reliant on raw resources, such organic arable farming (Stabenau et al., 2018). Harvesting nuisance aquatic plants can also be cost effective compared to other in-lake management schemes, e.g., by reducing the P load by as much as 50%. (Bartodziej et al., 2017). Focusing on harvesting existing invasive plants may therefore offer a sustainable option for joint freshwater habitat improvement and resource recovery schemes (Carson et al., 2018). Caution must be exercised in these cases and a risk assessment must be carried out before harvesting invasive plants to determine the likelihood of facilitating further invasion elsewhere by unintendedly transporting propagules, indeed harvesting some invasive species may also be illegal (Fletcher et al., 2020).

One difficulty of scaling up phytoextraction is the transporting and processing of material beyond the site, while assessing the costs and benefits can prove challenging (Edgar et al., 2021). Therefore, using the harvested biomass close to its source will help build circular systems within multi-functional landscapes, and is certainly the most promising application of this approach (Atkinson et al., 2014). For example, ideal locations for targeted plant harvesting and re-use could be agricultural lands adjacent to river navigations or recreational waterbodies where existing nuisance plants are periodically removed and deposited at the water side, resulting in nutrients leaching back into the waterway (Boerema et al., 2014). Here it has been highlighted that the different plant community types and harvest regimes that can facilitate the removal of pollutants from waterways. Catchment managers and stakeholders can therefore consider phytoextraction as part of a wider suite of measures that target the sources, mobilisation, delivery, and impacts of pollution. This will help to build circular economy approaches back into existing land management.

5. Conclusion

Phytoextraction using macrophytes is a nature-based solution to the dual problem of water pollution and declining supplies of raw materials. Targeting phytoremediators that can be harvested from existing freshwaters or for deliberate planting (and subsequent harvesting) can be challenging because of the different approaches to quantifying phytoextraction potential. By examining the relationship between biomass and tissue concentrations on standing stocks of macronutrient- and micronutrient-type pollutants the influence of these two components have been disentangled. High biomass plant communities, particularly those comprised of competitor or large stress tolerator species, will produce greater standing stocks of macronutrients (such as N and P) for harvesting. Yields of micronutrient-type pollutants are more likely to be influenced by their tissue concentrations (which are determined by plant physiological traits) and targeting these communities for phytoextraction using species-specific knowledge of hyperaccumulation potential is more appropriate than community generalisations. However, by utilising a high frequency harvest regime over a multi-annual time scale ruderals yield far greater amounts of macronutrients and micronutrients for export than competitor or stress tolerator dominated communities. These results can help guide environmental managers in their approach to developing circular economy schemes to improve water quality and export nutrients and emphasise the need to consider both the plant community and the harvesting regime employed.

Acknowledgements

Funding for this work was provided by the Scottish Government Hydro Nation Scholars Programme.

Appendix A. Supplementary Information

Supplementary material related to this article can be found online at https://doi.org/10.1016/j.resenv.2022.100050.

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