Riverflies and Riverflows: Effectiveness of Citizen Science for Monitoring Freshwater Ecosystems in the Hope Valley

Abstract

Understanding and protecting freshwater ecosystems and the services that they provide requires monitoring methods which capture both structural and functional components of the ecosystem. Macroinvertebrates are commonly sampled as one method of determining water quality due to their role in fish diet and status as an indicator of pollution, but data can be time consuming and expensive to collect. The Angler's Riverfly Monitoring Initiative engages recreational fishermen in recording the abundance of benthic macroinvertebrate larvae in order to reduce the expense of data collection and increase public engagement in science and nature. In this report, amateur data is compared to more thorough laboratory analyses and the ability of angler's RIverfly scores to determine changes in fish populations is examined. The Riverfly monitoring scheme data is not strictly comparable to scientific analysis in assessing macroinvertebrate communities; however relative trends in Riverflies are represented. Monthly Riverfly score does not correlate with average fish catch per hour for either trout or rainbow trout, but Riverfly score and grayling catch per hour are positively associated, providing a useful way of predicting changes in ecosystem service provisioning.

Keywords: *Biomonitoring, Ecoystem services, Freshwater Fisheries*

Introduction

Freshwater ecosystems cover only 0.8% of the world's surface but make up almost 6% of described species (Dudgeon *et al.,* 2006). This biodiversity supports a wide range of ecosystem services; regulating services such as flood and water quality regulation, cultural services such as education and tourism as well as provisioning of water and fish (Brown *et al.,* 2012). Recreational fishing in freshwater ecosystems provides £1bn of UK household annual income through rod sales and tourism as well as encouraging physical activity, mental well-being and conservation of the surrounding areas (Brown *et al.,* 2012).

However, due to numerous stressors, freshwater ecosystems are among the most endangered in the world, with declines in biodiversity far exceeding any of the most affected terrestrial ecosystems (Sala *et al.,* 2000). Freshwater habitats face threats from invasive species, over-exploitation, habitat degradation, flow modification, and water pollution (eg. Dudgeon *et al.,* 2006), all of which contribute to an overall decline in biodiversity and ecological quality. In the UK, more than 350 serious or significant water pollution events are reported each year to the Environment Agency due to domestic sewage, urban run-off, industrial effluents and farm wastes (EA, 2013). During the decomposition of such pollutants, dissolved oxygen in the water may be used up at a greater pace than it can be replenished, causing reduced fitness and even asphyxiation of stream biota. Organic effluents can also reduce light available to photosynthetic organisms (Lenntech, 2015). Reduced water quality also results from erosion of riverbanks due to grazing and trampling, which limits oxygen available to aquatic life as well as potentially ruining plant, invertebrate and fish spawning habitats (Herbst, 2005; Giles *et al.* 2004).

In order to understand how ecosystems are changing in response to stressors and manage sites accordingly, a monitoring system which reflects ecosystem interactions and processes is vital (Friberg *et al.,* 2011; Young *et al., 2014*). In freshwater ecosystems, biomonitoring using macroinvertebrates has dominated assessments of environmental quality due to their ubiquity, well known taxonomy and sensitivity to a wide range of stressors, particularly organic pollution (Rosenberg and Resh, 1993; Friberg *et al.* 2011). Aquatic macroinvertebrates have roles in key ecosystem processes such as litter decomposition and are an important component of fish and bird diets (Herbst, 2005), so understanding changes in macroinvertebrate composition can help conclusions about the status of the wider community to be drawn. Such assessments can be more useful than chemical analyses,

especially for identifying non-point pollution (for example sediment washed from ploughed land or from fresh ditching work (Giles *et al.* 2004)) which may be missed in a single water sample (Herbst, 2005). Stream invertebrates are directly affected by everything that flows over them, giving a good indicator of the quality of a water body (Rosenberg and Resh, 1993; Herbst, 2005).

One way in which this data can be collected is through Citizen Science; recruiting non-scientists to collect data on a voluntary basis as a part of their everyday routines or activities. This has been recognised as a valuable way to collect biomonitoring datasets covering large spatial and temporal extents at a minimal cost (Table 1) (Kaartinen *et al.,* 2013; DeVictor *et al.* 2010; Gallo, 2011). Due to the high number of stakeholders who compete for freshwater resources (Dudgeon *et al.* 2006), freshwater habitats in particular may benefit from a sense of stewardship from local inhabitants (Winfield, 2014). The Angler's Riverfly Monitoring Initiative (ARMI) encourages anglers already involved in the welfare of the river to take monthly 3 minute kick samples and record the abundances of 8 main macroinvertebrate groups (Table 2). This is then translated into a Riverfly score for that month based on a simplified form of Biological Monitoring Working Party (BMWP) scores (The Riverfly Partnership, 2015a). Citizen Science

schemes such as this also offer an educational benefit to participants and can be valuable in reconnecting people to nature more generally, which can itself prevent further damage to ecosystems by increasing environmental awareness of participants (DeVictor *et al.,* 2010). Increased monitoring also acts as a deterrent to potential polluters (The Riverfly Partnership, 2015b).

Table 1. Attributes of scientific investigations which make them suitable for Citizen Science schemes (Gommerman and Monroe, 2012)

Attribute	True for the Riverfly project?
Labour intensive data collection	Yes
Field data collection	Yes
Quantitative	Yes
Well designed, easy protocol	?
Broad spatial/temporal extent	Yes
Internet accessible data submission	Yes
Guide materials	Yes
Professional assistance available	Yes
Large data sets needed	Yes

When training and education materials are provided, the information obtained by Citizen

Science has been found to be largely comparable to data collected by scientists (Gommerman and Monroe 2012). However, Citizen Science projects have also been criticised for being non-scientific and lacking verification (Gallo, 2011), so studies assessing their ability to provide reliable information are vital (Tregido *et al.*, 2013). Despite the widespread use of the Riverfly project, since its launch in 2007 there has been very little discussion of its success. In this report, an attempt is made at assessing the effectiveness of the Anglers' Riverfly Monitoring Initiative in providing reliable water quality information to contribute to management decisions.

The first question that this report addresses is the ability of the project to measure macroinvertebrate communities. Onsite identification tends to be less effective than standard procedure when in a lab (Cao *et al.,* 2003), and the abundance grades awarded by the anglers are just estimates, so there may conceivably be some discrepancies between scientific and amateur data. However, in order for the data collected to be considered reliable as a basis for future decision-making, the data generated should be largely comparable to that collected by a more thorough scientific approach (Roy *et al.,* 2012).

Therefore, in this report a comparison of my own samples is made to Riverfly samples taken by the Peak Forest Angling Club at 2 sites on the River Noe in order to determine the effectiveness of amateur sampling and to examine whether the project data gives a representative view of macroinvertebrate communities. The PFAC estimates for macroinvertebrates from a total of four sites are also compared to my counts from the same given sample to assess the comparability of processing.

Table 2. The 8 groups of macroinvertebrates which are recorded by anglers participating in the Riverfly monitoring scheme.

Riverfly groups
Cased caddis
Caseless caddis
Mayfly (Ephemeridae)
Black-winged olive (Ephemerellidae)
Flat bodied (Heptageniidae)
Olives (Baetidae)
Stoneflies
Freshwater shrimp

The aims of the Angler's Riverfly Monitoring project are to protect river water quality, further understanding of Riverfly habitats and furthermore to actively conserve those habitats (The Riverfly Partnership, 2015a). However, as yet, little attention has been given to how the information gained from their research could support a broader ecosystem approach to monitoring. Discussion of ecosystem services monitoring within fisheries

is limited (Daily, 2000), despite recognition of the need for indicators which can be directly linked to ecosystem conditions (Hilty and Merenlender, 2000; Carpenter *et al.,* 2009; Parliamentary Office of Science and Technology, 2007).

Given this concern, the second question that this report investigates is whether Riverflies are a good indicator of ecosystem service provision in the form of fish. Despite being sensitive to many of the same stressors which affect fish and constituting a large part of fish diet, (Giles *et al.,* 2004) the links between macroinvertebrates and fish population abundances are not well understood (Bryce, 2014). In the second part of this report, the relationship between fish catch and Riverfly index is explored as a valuable way of predicting changes in service provisioning.

Methods

Study Area

The study was carried out on the River Noe in the Hope Valley in Derbyshire, a tributary of the River Derwent which flows through the Forest of High Peak. The river's source is in the Edale Moors where it flows in an easterly direction through moorland and rural areas before reaching the villages of Hope and Shatton. A dense network of small, steep hillslope tributaries is closely associated with the river, along with several larger, more

significant sub-catchments including Peakshole Water and Bradwell Brook.

The River Noe is a heavily modified river which has been identified as unlikely to meet the 'good ecological status' required of water bodies by the European Water Framework Directive (EU WFD) due to disproportionate expense and technical unfeasibility (Crouch and Walker, 2013).

Coarse substrate and high energy flows (riffles) provide habitat for brown trout, bullhead and lamprey, whilst sequences of riffles and pools are ideal for grayling. The river is dominated by brown trout, with bullhead and grayling only accounting for a small proportion of the estimated biomass (EA, 2014). The river is also one of only three rivers in the UK where rainbow trout breed naturally (Card, 2011).

Several natural and artificial pressures exist on the river: water abstraction to support demand for public water supply from this area causes localised reductions in the quantity and velocity of flow as well as affecting sediment transfer, whilst run-off from surrounding land may disrupt water quality. Fish populations are also influenced by barriers to migration such as man-made weirs and a reservoir dam in the uppermost part of the catchment as well as exposure of natural steps in the bedrock under low flow conditions (EA, 2014). Bradwell Brook, Peakshole Water and the Upper Noe are

maintained as wild fisheries, whilst the lower river is lightly stocked. During March and October the river is not fished (Card, 2011).

1) Comparability of Amateur and Scientific Data: PFAC Kick Samples

Monthly kick sampling is carried out by members of the Peak Forest Angling Club at 4 sites on the Noe, based on instructions provided by the Riverfly Monitoring Project (Field Studies Council, 2007), excluding the 1 minute manual search which is advised. Six 30 second kick samples are carried out a few metres apart, incorporating different microhabitats. The contents of the net are emptied into a single bucket which is filled with water and poured by increments into a sorting tray. Macroinvertebrates are picked out using a plastic spoon or pipettes and sorted into 8 groups in a segmented tray. This process is done quickly, aiming to not miss more than 10% of animals. An estimate of the number of invertebrates in each segment of the tray is then recorded. From November to February, PFAC have transferred their sample to a container with 70% Industrial Methylated Spirits (IMS) and sent it back to the lab for analysis. Only a handful of grit from the kick sample is included in the container with the sample, with the rest being returned to the river without analysis. Due to the voluntary nature of the project, samples from all

monitoring sites for all months are not available.

My Samples

On the 25th February, six 30 second kick samples were taken at monitoring sites 1 and 2, which had the fullest data sets available for comparison. Each 30 second sample was taken a few metres apart, incorporating different microhabitats and was stored in an individual pot in 70% IMS. Water samples were also taken which were analysed for nitrate, nitrite, phosphate, and ammonia concentration using a Palin test kit. pH measurements, dissolved oxygen and conductivity were also taken at each point before the kick samples were carried out to avoid disturbing the water before sampling.

Sample analysis

Samples were sieved and macroinvertebrates were picked out from a tray under a bright light. Each individual was identified to family level using Dobson *et al.* (2012).

All procedures were carried out following the procedure outlined in the Environment Agency's standard sampling and analysis manual (Murray-Bligh, 1999), excluding the one minute manual search which is usually performed after kick sampling.

Statistical analysis

Variation between sites in diversity, macroinvertebrates and Riverfly scores from my counts and from PFAC counts was tested

using ANOVA in order to determine how sensitive the Riverfly score is to differences in habitat and how potential Riverfly stressors vary between sites. Water chemistry data was also compared between sites with ANOVA to provide potential explanations for variance in macroinvertebrate abundance. My counts of macroinvertebrates from PFAC samples were compared to PFAC estimates recorded on site using paired t-tests.

2) Linking Riverfly scores to fish catches

Catch per unit effort provides a reasonable index of actual abundance (Giles et al., 2004). Therefore, fish catch data from the River Noe from February 2013 to February 2015 has been collated, specifying the fish caught per hour for each angler visit made to the river. The river is divided into beats to ensure fishing effort is distributed along the whole river and to avoid overfishing in certain areas. Beats 4 and 6 are not fished, which is where monitoring sites one and three lie (Table 3), meaning that the Riverfly data for these sites cannot be linked to fish catches. Given that Riverfly data from monitoring site 4 is also quite patchy, this leaves only a small data set for comparison. However there are no significant differences in either Riverfly score (ANOVA, F <0.01, $d.f.$ = 1,70, p = 0.99) or fish catch per hour (ANOVA, F = 0.3167, $d.f.$ = 1,534, p = 0.5738) between sites and beats respectively. Therefore, data has been combined across sites to give an

average Riverfly score and an average fish catch per hour for each month. Linear regression has been carried out to determine the relationship between Riverfly and fish catch per hour, which is repeated for individual fish species and different Riverfly groups to establish the most important determinants of the link.

Table 3. Riverfly Monitoring sites and the Beats where they are located

Monitoring site	Beat	Fished?
1	6	No
2	7	Yes
3	4	No
4	2	Yes

Results

Variation Between Sites

All three sampling sites were at an altitude of 160m. and within 7.5 km. of one another. The river at these sites was 6-8 m. wide and less than half a metre deep. Boulders were the dominant substrate at each site and the water velocity was 20-30 m^3/s (Table 5). There was significant inter-site variation in conductivity, nitrate concentration and pH (ANOVA F > 7.6, $d.f.$ = 1, $\geq$9, p < 0.05), with Site 1 having a higher conductivity and nitrate concentration than Site 2, but a lower pH (Table 4).

From PFAC counts of all four sites it is clear that there are differences in macroinvertebrate communities, with sites 1 and 3 demonstrating a greater diversity (Figure 1). That being said, overall diversity did not differ significantly between sites (ANOVA, F = 1.407, $d.f.$ = 1,4, p = 0.3011). Mayflies, stoneflies and caddis flies were present at all sites, although mayflies were the only order which were present at every site for every month since February 2013. Site 1 had a significantly higher abundance of stoneflies than Site 4 (ANOVA, F = 2.493, $d.f$ = 3,59, p = 0.0438439). Shrimps were present in only small numbers (<15) at all sites apart from Site 3 which had a significantly higher abundance than all other sites (F = 18.67, $d.f.$ = 3,59, p <0.01), with a significant interaction of month. Site 3 also had a higher abundance of caddis fly larvae than both sites 1 and 2 (ANOVA, F = 4.6, $d.f.$ = 3,59, p <0.05). These differences were small though and there were no significant differences in Riverfly score between sites for either my score (ANOVA, F = 0.39, $d.f.$ =1, 12, p = 0.7) or for the PFAC scores since 2013 (ANOVA, F <0.01, $d.f.$ = 1,70, p = 0.99)) with no significant interaction of month. During February 2013 to February 2015, fish catch per hour also didn't vary between beats (F = 0.3167, $d.f.$ = 1,534, p = 0.5738).

Table 4. Water characteristics for sites 1 and 2 with standard deviation in brackets. Values followed by a different letter differ significantly.

Site	Altitude (m)	Slope (m/km)	Velocity (m³/s)	Distance from source (km)	Mean width (m)	Mean depth (m)	Dominant substratum particle size
1	160	8.90	30.0	1.3	6	47.25	Boulder
2	160	1.25	24.5	6.0	8	16.00	Boulder

Table 5. Environmental data for sites 1 and 2

te	Conductivity (s/m)	pH	Amonnia (mg/l)	Nitrate (mg/l)	Nitrite (mg/l)	phosphate (mg/l)
1	393.5 (0.1)a	8.0 (0.02)a	0.01 (0.006)	2.43 (0.23)a	0.01 (0.01)	0.10 (0.05)
2	137.3 (0.3)b	8.2 (0.03)b	0.02 (0)	1.01 (0.20)b	0.00475 (0.003)	0.08 (0.05)

## 1) The effectiveness of the Angler's Riverfly monitoring project for assessing macroinvertebrate communities.

a) Comparability of sampling

A comparison of PFAC's 3 minute kick sample to the three minute kick sample that I retrieved from the two sites in February revealed noticeable differences in the invertebrates captured by the sampling process. Samples differed in diversity (Figure 1), abundance (Figure 3a, b) and family richness (Figure 3c). For Site 1, there were 12 families which were found in my samples but not the PFAC samples and 2 families which were found in their samples but not mine. At Site 2, there were 6 families which were found in my sample but not the PFAC's and 1 which was found in their sample but not mine (Table 7) all of which were present in fairly low abundances (<15). As I only took one kick sample the statistical significance of these differences can't be tested, but both Site 1 and Site 2 revealed a consistently lower abundance and diversity across all taxa for PFAC samples than for mine. These differences translate into higher Riverfly scores for my sample for the same month (Table 6), despite the samples being taken only 3 days apart. The relative trends in Riverfly populations between sites are reflected in all measurements- all scores suggest a slightly higher Riverfly score at Site 1 than at Site 2 (Table 6).

Table 6. Comparison of PFAC Riverfly scores from February samples

	Riverfly score from PFAC count of PFAC sample.	Riverfly score from my count of a PFAC sample	Riverfly score from my sample
Site 1	8	9	15
Site 2	6	8	12
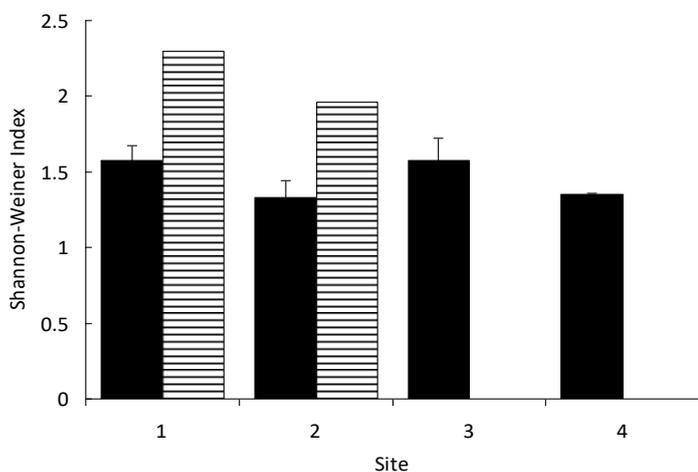
Figure 1. Shannon-Weiner Diversity Index for sites 1-4 from (■) my average counts of November-February PFAC samples and (▤) my February samples. Error bars for PFAC samples show standard error of the mean.
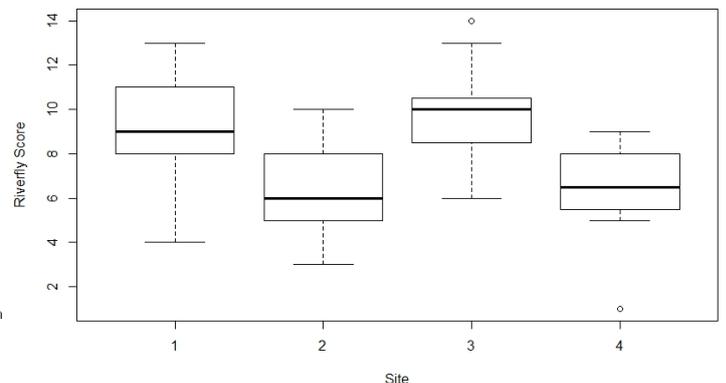
**Figure 2. Varaition in Riverfly score at sites 1-4 since 2013.**

b) Comparability of processing

There was a large amount of variation in the comparability of my counts of PFAC samples and PFAC estimates of the same sample. Sometimes differences were consistent, for example at Site 2 Heptageniidae were consistently underestimated by PFAC whereas Baetidae were always overestimated, whilst others were less consistent over time and between sites. One noticeable difference was a count of only one cased caddis where I recorded 26 for Site 1's November sample. The differences in counts were significant for cased caddis (paired t-test, $t = 4.0$, $d.f. =12$, $p < 0.05$) and Baetidae (paired t-test, $t = 2.5$, $d.f. = 12$, $p < 0.05$) (Figure 4). This translated into differences between my counts and PFAC estimates for Riverfly score in all but one of 13 samples (Figure 5). PFAC estimates were largely consistently conservative in comparison

December at Site 1 and January at Site 3 were considerably overestimated (Figure 5).

2) Linking Riverfly scores to fish catches

Overall, average fish catch per hour per month was not significantly correlated with average PFAC Riverfly score per month ($F = 1.95$, $d.f. = 1,19$, $p = 0.18$). Catches of trout and rainbow trout also didn't correlate with Riverfly score ($F > 0.4$, $d.f. = 1,19$, $p > 0.1$) (Figure 6). However, grayling were significantly positively associated with Riverfly score ($y = 0.0631x - 0.3463$, $F = 6.7$, $d.f. = 1,19$, $p = 0.01804$). From dividing the catches for the species further into those over 12" and those under 12", it was apparent that this was due to a strong positive correlation between large grayling and riverfly score (Figure 7) ($y = 0.0444x - 0.2532$, $F = 6.37$, $d.f = 1,19$, $p = 0.02064$) whilst for smaller grayling the relationship was not significant ($F = 2.0$, $d.f. = 1,19$, $p = 0.1736$).
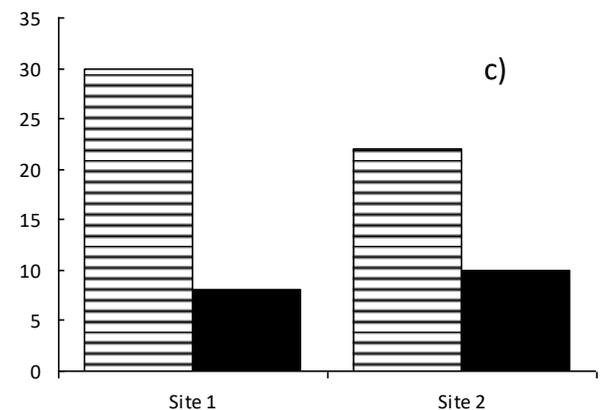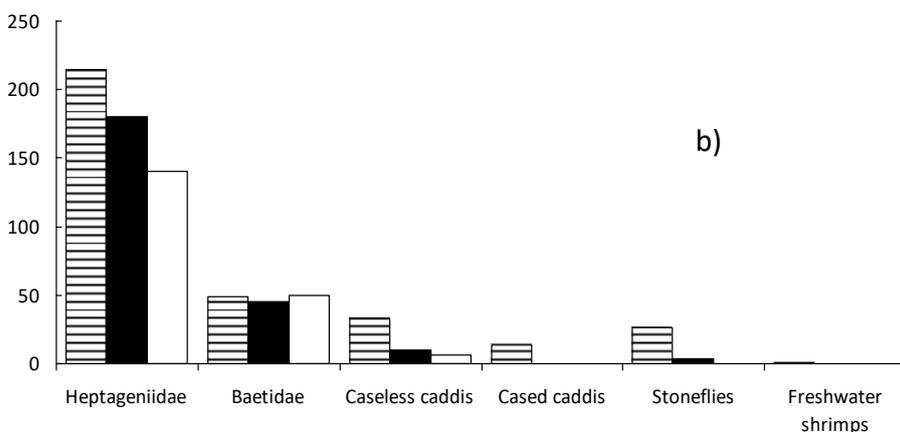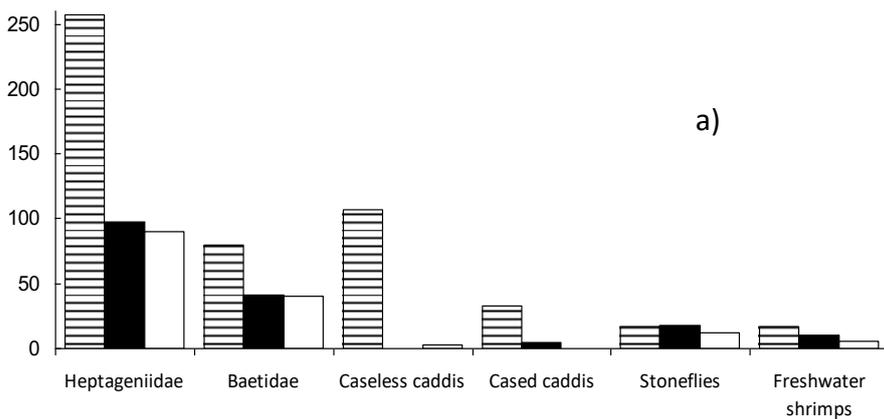
Figure 3. Abundance of Riverfly groups in (⊞) my sample (■) my counts of PFAC samples and (▢) PFAC estimates of PFACsamples from February for a) Site 1 and b) Site 2. Ephemeridae and ephemerellidae were not present at eitherof the sites so were excluded. c) shows family richness for sites 1 and 2. PFAC estimates of PFAC samples are not available for family richness as only order level data is included in counts.

None of the individual macroinvertebrate groups (freshwater shrimps, mayflies, stoneflies and caddis) averaged across sites per month correlated with average fish catch for the month ($F > 0.08$, $d.f. = 1,19$, $p > 0.1$). Likewise, no single averaged macroinvertebrate count correlated with grayling catch alone ($F > 0.0008$, $d.f. = 1,19$, $p > 0.05$) although mayflies showed a strong link ($F = 4.1$, $d.f. = 1,19$, $p = 0.057$
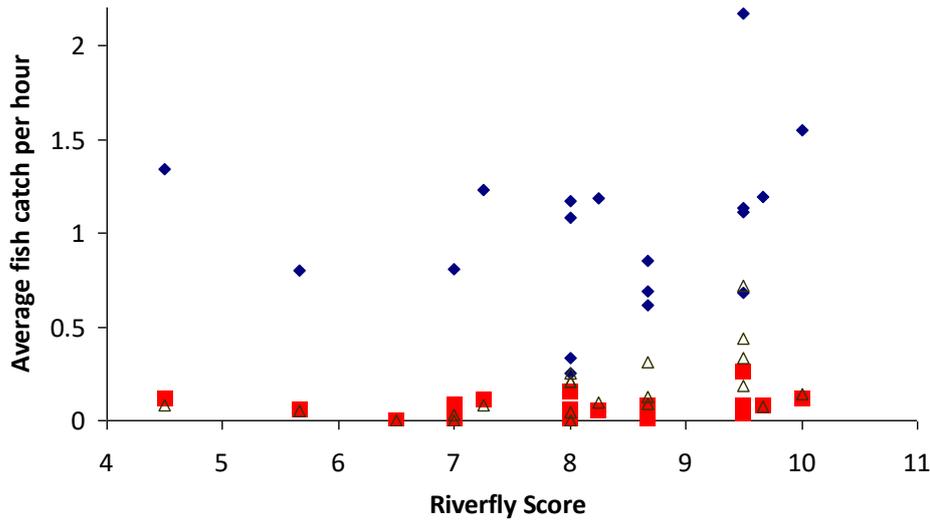

Figure 6. Relationship between monthly Riverfly score averaged across sites and average fish catch per hour per month.

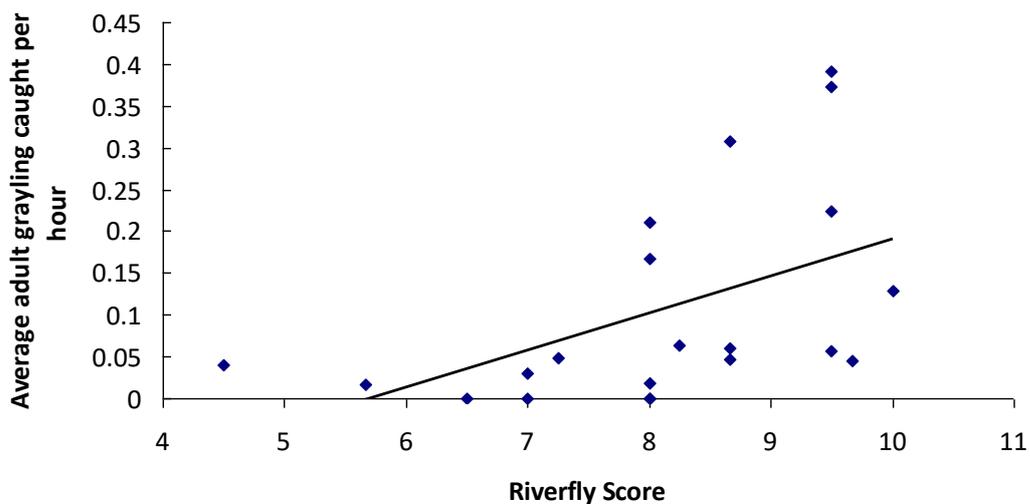◆ **Shows brown trout** ■ **Shows rainbow trout** △ **Shows grayling**

Figure 7. Relationship between monthly Riverfly score averaged across sites and average adult grayling caught per hour per month.

Discussion

This report has shown that Citizen Science in the form of the Angler's Riverfly Monitoring Project can be used to gain assessments of Riverfly populations and has the potential to provide information about ecosystem service provision in the form of fish.

There is a great deal of variation in the data between months and sites, which is common in macroinvertebrate samples; one- third of total variation between sites and times in a water body can be assigned to normal background variation between replicate samples taken at the same site on the same day, regardless of the type or quality of a sampling site (Clarke, 2009). On the River Noe, site clearly plays a role in determining Riverfly abundance but is not significant. Both my samples and PFAC samples reflect a lower Riverfly score at Site 2, which suggests the differences between sites are unlikely to be due to different PFAC members taking samples at different sites.

Lack of significant difference between sites despite considerably different habitats could be due to the coarse measurement that the monitoring project provides, categorising the invertebrates too broadly for differences to be noticed. Indeed, Tregido *et al. (*2013) found that the simplified methodology used in citizen science reduced the ability of monitoring to detect subtle changes in the variable being measured, which could certainly be an issue for the Angler's Riverfly Monitoring Initiative. There is no trend in Riverfly scores over the 2 years it has been measured on the River Noe, yet during this time ephemerellidae have become very rare where they used to exist at all sites in abundances greater than ten. This is an area which is not looked into here but an important observation that analysis of ARMI data provides. It highlights that the scores, however, are quite a blunt measurement of macroinvertebrate communities which may also hinder water quality assessment.``` The lowest level of identification, that is species identification, provides the best indicator for water quality assessment, although family level is a good alternative (Feio *et al.* 2011). The higher level classification used by the Riverfly monitoring project (order level in some cases) may be too coarse a measure to notice subtle changes, only detecting severe declines in water quality. However, the frequency of sampling undertaken by PFAC means that variation due to pollution events or other significant stressors affecting Riverfly populations are more likely to be picked up than a more thorough RIVPACS analysis which involves only two seasonal samples per year. Further information could help identify how far away sites should be to minimise sampling effort (Clarke *et al.,* 2009).

Samples taken by the PFAC and my own samples did not seem directly comparable in either diversity or overall Riverfly score, which is likely to be due to the fact that the PFAC don't examine gravel at the bottom of the bucket for macroinvertebrates. The Riverfly Partnership does recommend washing the stones picked up in sampling by filling the bucket with water from the river and agitating the sediment (FSC, 2007), although this can be challenging given how well adapted many of the mayfly families are to clinging on to stones in fast moving water (Chinnery 1993). Consistency between months and sites is likely to be more important than a precise sampling technique though; as long as sampling is carried out in the same way each time, deviations from the norm can still be reliably identified (Clarke, 2009). The three minute kick sample which has become commonplace for assessing macroinvertebrate communities captures only around 50% of species and 60% of families found in 6 replicate samples (Furse *et al.,* 1995); samples are to get an idea of the relative population size through time rather than an exact value. Table 7 shows that families which are missed from PFAC samples but found in mine tend to be of the orders Diptera and Mollusca, which are not measured under the Riverfly project and so are unlikely to make a difference to the assessment of river quality gained.

Differences in processing between amateurs in the field and more thorough scientific processing in the lab are also apparent in assessment of the samples. Some of the variation could be due to misidentification, for example in the case of the November Site 1 sample, I counted 26 cased caddis whilst the PFAC counted only one. Other discrepancies could be due to the challenges of fieldwork; animals are counted whilst still moving around at the side of the river. Given that most of the macroinvertebrates concerned in this study are less than 1 cm long, these counts could be difficult, so perhaps this level of variation is to be expected.

Apparent differences in Riverfly score due to slight variations in sampling technique suggest that overall Riverfly score may not be comparable with scores from other catchment areas, which in turn makes determining the precise significance of scores challenging, particularly for the Peak Forest Angling club where trigger and predicted levels have not yet been determined. Without the ability to compare the scores to other catchments or reference sites, the score means very little and drivers for the change still need to be identified.

Environmental data is often advanced as a possible means to identify reasons for change in macroinvertebrate populations (Herbst, 2005). In this study, however, there was very

little evidence for such characteristics having a role in determining macroinvertebrate abundance; whilst there were significant differences in pH, conductivity and nitrate concentration at Site 1, there was no corresponding difference in Riverfly diversity or Riverfly score or even in the abundance of individual groups. Although water chemistry information from sites 3 and 4 is missing due to constraints of fieldwork, this highlights that static traits such as water chemistry may be a poor predictor of trends in other components of the ecosystem. This is becoming increasingly recognised; for example under the Water Framework Directive chemical analysis makes up only a small part of the various measurements to identify the state of a water body, as healthy ecosystems depend on more than just good water quality and physical habitat (European Commission, 2002).

It could be that the characteristics measured here are not an important determinant of Riverfly abundance; all the water quality measurements taken were within the EU recommended water quality parameters for salmonoid rivers, although the toxicity of some chemicals to macroinvertebrates can vary with factors such as chloride concentration (Giles *et al.,* 2004). The Salmon and Trout Association fly survey (Hayes, 2008) found that the dominant factor affecting Riverfly decline is low flows which can lead to decreased oxygen

and lower dilution of pollutants. Heavy metal concentrations could also be a useful measurement to obtain; the River Noe has failed to meet WFD standards for chromium levels and contains high quantities of zinc in the lower River from non-moorland land management (Crouch and Walker 2013) which may help to identify drivers for Riverfly population change and support the current information being collated.

Another factor which could influence comparison between sites is the 1 minute manual search outside the 3 minute kick sample which is advised by the Riverfly Partnership (FSC, 2007), but not carried out by the PFAC. Excluding the manual search has been found to result in significant changes to macroinvertebrate scoring systems such as Average Score Per Taxon (Bryce, 2014) and is therefore also likely to affect Riverfly scores.

Other Citizen Science projects which face the difficulty of inconsistency between amateur collectors allow citizens only to collect samples which are then held for professional analysis. For example, *Florida* LAKEWATCH involves volunteers in collecting water and algae samples which are then analysed by the University of Florida's Fisheries and Aquatic Sciences water chemistry laboratory (Florida Lakewatch, 2015). For the Invaders of Texas programme, photos are collected which are verified by experts (Invaders of Texas, 2015).

Riverfly data doesn't really lend itself to this methodology, as counting the insects is the time-consuming part of the monitoring and identification can't be made from a photograph. That being said, spot-checks from professionals may help to standardize the sampling technique. Contact between professionals and volunteers also produces higher quality data as citizens feel, through their engagement with an actual researcher in the professional sphere, that they are making a genuine and more important contribution (Nerbonne and Nelson, 2008).

Citizen Science should allow the study of different ecological levels from individual species to ecosystem processes (Maltby, 2003), but currently there are few ecosystem level analyses (Kaartinen, 2013), with limited further study of Citizen Science data to answer experimental questions beyond monitoring (Gallo, 2011). Biomonitoring efforts often lack support from ecological theory, with metrics that don't always link to useful ways of determining changes in ecosystem state (Friberg *et al.* 2011). Therefore identifying clear links between structural ecosystem components such as Riverflies and ecosystem services could be extremely useful.

There is no relationship between overall average fish catch per hour per month and average Riverfly score across the 4 sites for that month, which could be for several reasons. Firstly, disturbance often wipes out invertebrate families, but has only indirect effects on fish (Berkman *et al.*, 1986) so the link between the two may not seen when fish catch is being measured. Also, the river is stocked yearly with brown trout, which may ameliorate any effects of decreased Riverfly abundance on fish populations.

The catch of grayling over 12" is significantly positively associated with Riverfly Score. Although this could be an artefact of a small sample size, it could also be because adult grayling rarely feed on the surface, whilst brown trout and younger grayling are frequent surface feeders (The Wild Trout trust, 2015). This means that the adult grayling are likely to be more reliant on benthic macroinvertebrate larvae than terrestrial insects and are therefore are more tightly coupled. If this hypothesis is correct, it suggests the most important aspect of the link between Riverflies and fish is their role in fish diet, rather than their role as an indicator of pollution. Low levels of organic pollution have been shown to have positive effects on fish (Aas et al, 2011) but they are very sensitive to depleted oxygen, a sign of organic pollution (EA,2011) which makes this finding unexpected.

Despite suitable habitat and water chemistry in the tolerated range (EA, 2014), grayling populations have declined since 1999 on the River Noe so understanding the links between

readily available Riverfly data and grayling abundance is vital. Comparison of other functional indicators may also be important (Mandelik *et al.,* 2012); measures such as leaf litter decomposition may provide less variable, higher sensitivity data than Riverfly scores (Young *et al.,* 2004) that can be more easily linked to stressors (Gessner and Chauvet , 2002). Such measures may not be ideally suited to Citizen Science projects though as they are difficult to explain (Gessner and Chauvet, 2002).

Whilst Riverfly scoring may not be perfect for predicting a range of ecosystem services, as a Citizen Science scheme it does offer an ecosystem service in itself (Franco, 2013). A methodology of this kind encourages people to be active and get involved in the study and conservation of their environment, which promotes mental and physical well-being (Brown *et al.,* 2012). However, as with many Citizen Science projects, the demographic for the Riverfly partnership at the club is a mature membership; participants are educated people least in need of developing scientific understanding and environmental awareness or skill (Trumull *et al,* 2000). Involvement with local schools and Universities, an initiative already being practised at the Club, could encourage the benefits of Citizen Science to reach a broader range of people. This could also improve the quality of decisions and

recommendations concerning river management in future by incorporating more voices into the discussion, representing a broader range of opinions and hopefully encouraging initiative from within the community (Gommerman and Monroe 2012, Ellis and Waterton, 2004).

It is important that the questions being asked of Citizen Science projects are identified (DeVictor *et al.*, 2010). Continued involvement in Citizen Science is known to be higher in projects where there is access to initial results and information about how the data gained is being used (Gommerman 2001, Silverton, 2009), a feature that may be lacking from the current project. Additionally, as no trigger levels have been set, data has not yet actually been put to use and participants can feel like they're doing a bad job if they don't notice anything change in their region (Gommerman and Monroe, 2012).

Conclusion

Macroinvertebrate sampling through Citizen Science provides an easy and efficient way to monitor water quality. Whilst the data obtained is not directly comparable to more thorough scientific analyses, the relative trends in macroinvertebrate communities between sites and months are represented. Citizen Science data is rarely used for further investigations, but there is opportunity for Riverfly data to be linked to information about

the wider ecosystem and in particular ecosystem services in the form of fish populations. This report showed that for at least one species of fish, Riverfly score can be used to predict fish abundance. More information about the determinants of the role of Riverflies in fish populations could help to justify further conclusions about ecosystem state from Riverfly scores and help identify stressors. Citizen Science projects also provide benefits to individuals and communities as well as the habitats that the schemes are based in.

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