UCL DEPARTMENT OF GEOGRAPHY

Determining annual, seasonal and microhabitat variation in invertebrate and microbial decomposition rates using colonisation traps: A citizen science tool

Katherine Keates

Supervised by Steve Brooks

August 2018

This research dissertation is submitted to the MSc Aquatic Science at University College London.

DEPARTMENT OF GEOGRAPHY

M.Sc. in Aquatic Science

Please complete the following declaration and hand this form in with your M.Sc. Research Project.

I,.....Katherine Keates.....

hereby declare :

(a) that this M.Sc. Project is my own original work and that all source material used is acknowledged therein;

(b) that it has been prepared specially for the MSc in Aquatic Science of University College London;

(c) that it does not contain any material previously submitted to the Examiners of this or any other University, or any material previously submitted for any other examination.

Signed : .....Katherine Keates.....

Date : .....30/08/18.....

Abstract

The number of citizen science projects within environmental areas have increased in the last 20 years following increased public awareness in pollution and endangered species. It provides a low-cost platform for large scale data collection on a global scale as well as enabling local community engagement. Monitoring of river water quality through biotic indicators is an ongoing program co-ordinated by The Riverfly Partnership, with trained volunteers performing kick-samples and generating ARMI scores. Functional indicators including leaf litter decomposition rates are also considered important to monitor water quality.. A potential citizen science tool is using colonisation traps to measure decomposition rates and invertebrates. This project aims to evaluate the use of colonisation traps and further understand the use of decomposition rates to monitor water quality in rivers. Colonisation traps were placed in eight sites along the River Mimram, with coarse and fine mesh bags. The traps were left for two to three weeks in the river to enable invertebrate and microbial decomposition rates to be generated. The colonisation traps were also used to measure invertebrate composition A MoRPh physical habitat survey was also completed to determine variation in habitat complexity.

There was no significant difference between annual invertebrate and microbial decomposition rates. Variation was seen between sites in invertebrate abundance and richness. There were differences in functional feeding group (FFG) distributions. Seasonal variation showed significant difference between both invertebrate and microbial decomposition rates (summer higher than spring). FFG distributions showed higher prevalence of shredders in the summer and less collector-filterers. Several physical habitat parameters were corelated with biotic parameters such as invertebrate abundance. Microhabitat variation only showed significant differences in microbial decomposition rates and invertebrate abundance with two microhabitats having distinct invertebrate communities from the rest. Invertebrate decomposition rates were found to be influenced by the bed material particle size and channel physical habitat complexity. Comparisons between colonisation traps and kick-sampling showed linear relationships.

Annual variations in invertebrate abundance due to high *Simuilum* abundances in 2018 and higher *Gammaridae* abundances in 2017. Seasonal variation in invertebrate decomposition

rates could be linked to higher shredder % distributions. Microhabitat variation was seen, with the silt edge having significantly lower microbial decomposition rates to the fast channel in shade possibly linked to low channel physical habitat complexity. The colonisation traps are easy to use but there are some parts of the methods which are time-consuming and difficult for citizen scientists.

Number of words: 10400

Acknowledgements

I'd like to thank my supervisor Steve for driving me to the sites, helping put the traps into the river and with identifying the invertebrates. I'd like to thank the other volunteers who helped me survey the traps at the river especially Robin Cole who was a great help with the invertebrate identification. Also to The Riverfly Partnership for providing me with the River Mimram kick-sampling data.

Contents

Introduction

General introduction

Freshwater ecosystems make up 0.8% of the Earth's surface on Earth and contain 6% of described species, however they have one of the highest extinction rates compared to other ecosystems due to having many endemic species that rely on specific habitats which are being lost (Abell *et al.,* 2008). It is important, therefore, to continue to monitor and improve our knowledge of freshwater ecosystems to help conserve them and maintain their biodiversity. In the UK there have been initiatives set up to monitor rivers and lakes to better understand what healthy systems look like and what sources of pollution and other changing conditions are affecting them (Everard, 2008). This was very important in understanding the effects of acid rain pollution, and agricultural and industrial runoff (Batterbee *et al.,* 2008). However many of these issues, including new ones such as climate change, are still putting pressure on our freshwater ecosystems and it is therefore important to keep monitoring them (Watts, *et al.,* 2015). Citizen science is a major tool in collecting environmental data, allowing for large scale data sets to be obtained with comparatively small amounts of cost and time required to be taken up by government agencies, environmental charities or academic institutions (Roy *et al.,* 2012). There are several citizen science projects that have been set up to help conserve and monitor freshwater ecosystems in the UK. One important example is the Anglers' Riverfly Monitoring Initiative (ARMI) which assesses biological river health by sampling the aquatic invertebrate communities, focusing on riverfly families and allowing a score to be given to each river site which represents their biological health (Dunkley, 2018; Huddart, *et al.,* 2016). There are many other variables that are important to river ecosystem function that could be used to help assess biological quality alongside current monitoring efforts. In addition for long-term monitoring many of these variables are being measured through citizen science projects and schemes (Buytaert *et al.,* 2014). One of these parameters is the organic matter decomposition rates of the river, which is an important part of river ecosystem metabolism, recycling energy back into the system (Young, *et al.,* 2008). The use of colonisation traps which have been designed to measure decomposition rates, as well as provide another method of sampling invertebrates, could provide a new citizen science tool to allow for further monitoring of river biological health.,

This would also be another method to keep the general public engaged with the topic contributing to long-term success of monitoring (Roy *et al.,* 2012).

Citizen Science

The use of citizen science for collection of environmental data has allowed large datasets to be collected at regular intervals and at low costs for the government (Conrad & Hilchey, 2011). This is important in the current climate where monitoring projects are losing their funding and continued data collection relies on volunteers (Silvertown, 2009). Collecting this data is still vitally important in the continued monitoring and data collection of environmental parameters such as river ecosystem health which is a requirement under EU policy through the EU Water Framework Directive (Kallis & Butler, 2001). With the technological boom over the last 20 years citizen science projects have been able to fill this gap with increasingly easy ways to upload the data collected and to engage the public with citizen science opportunities through the internet and social media outlets (Dickinson *et al.,* 2012). The improvement in technical methods deployed in citizen science projects has also increased the reliability of data quality (Bonney, *et al.,* 2014).

There are many different types of citizen science projects that involve the collection of environmental data, ranging from identification of species images online to volunteers recording sightings of species such as the Big Garden Birdwatch or the National Bat Monitoring Programme (Roy *et al.,* 2012). Although there are many different types of citizen science projects they often follow a trend of focussing on important environmental issues in the news, as these build on the public's interest and thus may provide a larger response, which is a large driving force for citizen science projects (McKinley *et al.,* 2017). One topic in the news currently is plastic pollution in rivers and oceans. This has increased the awareness of the issue with increased beach/river cleansing, where plastic pollution is removed, but has also provided increased opportunities for data to be collected on a larger more consistent scale, so that the scope of plastic pollution and the main factors involved can be analysed (Syberg *et al.,* 2018). All these projects allow the general public to become involved, often with their local communities, and learn more about the environment while collecting data that could help improve it (Overdevest *et al.,* 2004; Tweddle *et al.,* 2012). Citizen science data

can form the basis of long term data collection to monitor changes in the environment on a large geographical scale (Magurran *et al.,* 2010). This can include surveying the local wildlife such as bumblebee nests or through the use of camera traps noting the presence of species and behaviours (Lye *et al.,* 2012; Steenweg *et al.,* 2016).

Previous and ongoing citizen science projects, focused on river health and ecology, often measure indicators of river health such as macro-invertebrate composition and water chemistry to determine water quality (Newman *et al.,* 2012). There are over 100 organisations across the UK using ARMI scores coordinated by The Riverfly Partnership including ZSL and Friends groups set up throughout London where groups of volunteers visit a number of sites along a river several times a year to sample the aquatic invertebrate community by kick-sampling (Dunkley, 2018). Other river monitoring projects focus on more specific species or conservation issues as well as monitoring the whole ecosystem, such as otter monitoring performed by volunteers in Northern California (Black, 2009). There are eight invertebrate families looked for within the river sample and if present these are given an ARMI score of 1-5 depending on their abundance (The Riverfly Partnership, na). These scores are then uploaded onto a national database so that changes can be seen and any problem sites which have scores below the trigger level can be identified and highlighted for government and independent agencies to rectify (The Riverfly Partnership, na). The methods used have been standardised and can be easily followed with little equipment needed, and training is offered to volunteers to improve data quality collection (Bennett *et al.,* 2011). Other, very similar, citizen science projects have been set up in this way for the general public to undertake in their own time instead of as part of a volunteer group/organisation specific session. The OPAL Water Survey project provided people who registered their interest with a starter kit which included a small net, sorting tray, sample tubes and the ARMI score identification guide (Rose *et al.,* 2016). Through projects like this people from all walks of life are able to come together and collaborate to collect information to improve the methodologies used to assess freshwater ecosystems to better help conserve and improve these ecosystems (Storey, 2016).

Monitoring river ecosystems

With the noticeable increase of citizen science projects being used in recent years, there are constantly new ideas being discussed for other areas in which citizen science can be used to

help in aquatic science research, and provide new opportunities for volunteers to learn more about our freshwater ecosystems and acquire the skills to help with their conservation (Biggs *et al.,* 2015). There are several variables which are looked at when defining the ecological status of a river which are used by environmental agencies, such as physical and chemical parameters including dissolved oxygen levels, phosphorus and flow rates (Griffiths, 2002). However biological indicators are also important in determining the health of a river, with the Water Framework Directive highlighting the main groups as macrophytes, invertebrates and fish with the species abundance and composition of these groups indicating river water quality (Everard, 2008). This is due to specific species being tolerant of different conditions indicating water quality and benthic macroinvertebrates have also been shown to be important and reliable indicators for ecological status (Iliopoulou-Georgudaki *et al.,* 2003).

Invertebrates as indicators of river health

Invertebrate composition varies depending on the type of river, the location within the river course and will vary within the different microhabitats within a river (in vegetation, on the river bank, on the river bed, and type of substrate) (Malmqvist, 2002). Invertebrates are also good indicators of river quality as they are found throughout the river system and many species vary in their tolerance to environmental stressors such as dissolved oxygen levels or they form pollution allowing for invertebrate species composition and richness (Clarke *et al.,* 2003). Macroinvertebrates are often sampled using the kick-sampling methodology allowing a representative sample of the invertebrate community found at a particular location to be determined (Beavan *et al.,* 2001). The standard method used is a 3 minute kick sample of all available microhabitats at a site by a hand net with a specific sized mesh allowing for consistency across data collection (Bennett *et al.,* 2011). However studies have shown that efficiency at sampling invertebrates can vary depending on whether the volunteers are trained or not, which could lead to data being seen as unreliable and not used by government bodies (Storey & Wright-Stow, 2017). Invertebrates are an important part of a river trophic structure as thy provide a large food source for many species higher up the food web and they can influence the environment around them (Wallace & Webster, 1996). The invertebrates can do this by filtering small particles out of the water and they can decompose organic matter recycling energy back into the system through carbon which is then transferred through food webs and downstream to other parts of the ecosystem (Kominoski

& Rosemond, 2012). Invertebrates have a wide variety of feeding habits which are split into five main groups. The percentage of each functional feeding group also varies depending on their location within the river and in different microhabitats (Cummins & Klug, 1979).The invertebrate group responsible for the majority of decomposition of organic matter is the shredders, which are often found in high numbers in the upper stretches of rivers feeding on organic matter from terrestrial leaf litter and decaying aquatic macrophytes within the river (Anderson & Sedell, 1979). As well as shredders, micro-organisms have also been found to play a role in allochthonous organic matter decomposition in small rivers and streams (Baldy *et al.,* 2002).

Leaf litter decomposition

Functional indicators such as organic matter decomposition have been considered alongside biological indicators as useful indicators for river health (Niyogi *et al.,* 2013). Leaf litter decomposition rates in particular can vary depending on many different environmental parameters, such as temperature and nutrient levels, and may be a more reliable indicator than other biotic variables that are currently being monitored (Pozo *et al.,* 2011). These different environmental and anthropogenic stressors can cause an increase or decrease in decomposition rates (Aristi *et al.,* 2012).

In many previous studies which measured river and lake decomposition rates, mesh bags containing leaves inside have been used to determine decomposition rates, determining the change in leaf mass overtime (Young *et al.,* 2008). With many rivers having extensive riparian vegetation especially in upper-reaches, the mesh bags would be a useful tool for citizen science projects looking at decomposition rates as they would be easy to collect (Webster, 2007). A citizen science project in the U.S uses leaves from local riparian trees in mesh bags to look at invertebrate composition in riffle beds (Leaf Pack Network, 2018). However this technique might not be as effective for citizen science projects looking specifically at leaf decomposition rates due to problems noted in many scientific studies, such as leaves easily fragmenting, making the process time consuming, as well as significant variation in leaf species chemistry and leaf quality affecting results (Tiegs *et al.,* 2007). Due to this, other methods have been used in scientific research when determining organic decomposition rates, such as cotton strips or wood debris (Aristi *et al.,* 2012). With invertebrates being largely responsible for the decomposition of allochthonous organic matter as well as their importance for other functions within the river ecosystem, it is important to sample the

invertebrate community alongside organic decomposition to help understand variations in decomposition rates and the other factors that may influence them (Anderson & Sedell, 1979; Pascoal *et al.,* 2003). This could be achieved by using colonisation traps which have been adapted the original method by placing fine and coarse mesh bags containing cloth paper as a leaf substitute into a structure that also acts as a means to sample invertebrate composition (Zhang, 2017). The use of these traps alongside kick-sampling and ARMI scores could allow for further understanding of the invertebrate community and functional indicators such as leaf decomposition and their use in determining river ecosystem health (Feio *et al.,* 2010). By widening citizen science opportunities and techniques this may keep volunteers interested and engaged in the environment, with more volunteers helping out and for more long-term periods (Roy *et al.,* 2012).

Colonisation traps

A previous study used the colonisation traps to compare decomposition rates and invertebrate composition between polluted and unpolluted rivers, as well as comparing the invertebrate community sampled in the traps to those of kick-samples already taken by citizen science volunteers (Zhang, 2017). From this study it was indicated that presence of high numbers of shredders such as *Gammarus* was linked to high decomposition rates which is consistent with the findings of other studies (Navel *et al.,* 2010). However there is still much unknown about how well the colonisation traps can be used to determine river health from invertebrate and micro-organism decomposition rates and invertebrate composition within the traps. The colonisation traps may also provide more information into how organic decomposition rates are affected by abiotic and biotic factors and there is further study required (Young, *et al.,* 2008). As well as this, for the colonisation traps to be effectively utilised as a tool for citizen science there must be an efficient standardised methodology implemented, allowing reliable data to be collected across all volunteer groups (Tulloch *et al.,* 2013). This means simple techniques that volunteers can be trained in, requiring little scientific equipment or allowing a scheme to be set up where samples can be sent to a lab for analysis as with water chemistry or eDNA samples (Biggs *et al.,* 2015).

Organic decomposition rates have been shown to increase in warmer temperatures and thus are higher in warmer seasons. Other factors such as leaf litter abundance and invertebrate abundance and composition are also seasonally variable and probably influence seasonal decomposition rates (Ferreira & Canhoto, 2014). With climate change resulting in warmer

temperatures and more severe weather events affecting many freshwater ecosystems and introducing more environmental stressors, there is likely to be annual variability in decomposition rates (Kominoski & Rosemond, 2012). This in turn causes shifts and changes in the biological make up of the system with increased vegetation cover in warmer summer months and changes in primary producers, zooplankton, invertebrate and fish species which rely on certain conditions (Power *et al.,* 2008). Therefore it is important to compare annual and seasonal change of river decomposition rates and invertebrate composition with the colonisation traps.

As well as this, river systems can have many different micro-habitats spread across the river length and in a single site., Slight changes in substrate, vegetation composition and cover, flow rate, position in river channel and presence of backwaters all provide a variety of habitats that support different invertebrate species compositions (Costa & Melo, 2008). With different invertebrate communities found in different micro-habitats along with different physical conditions relationships have been found between this and leaf decomposition rates (Kobayashi & Kagaya, 2005). As the colonisation traps are stationary when placed in the river, it is important to determine where they should be placed by citizen science volunteers to determine average site decomposition rates, if different microhabitats at a site affect decomposition rates and if so how.

By completing this project the aim is to further assess the current methodology behind the colonisation traps, including the field and lab aspects being appraised, and identifying improvements that could be made as well as any problems involved, so that colonisation traps could be used as an effective tool for citizen science in the future

Aims and Objectives

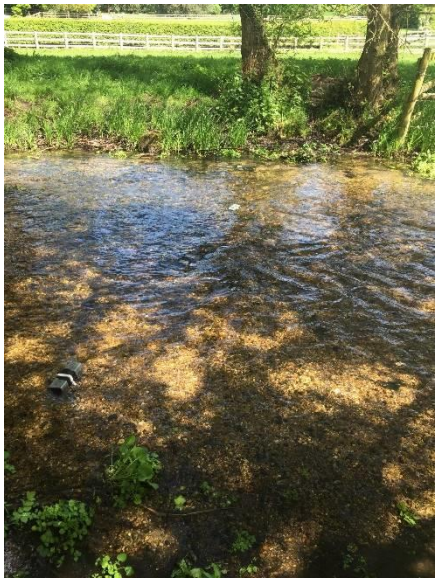
The main objectives of this study are to establish the effectiveness of colonisation traps as a method for determining decomposition rates and invertebrate compositions to be used as a tool for citizen science. The aims are to determine whether there is any variation in decomposition rates and invertebrate composition 1) annually, 2) seasonally and 3) between microhabitats, and if there are variations determining what could be influencing them by comparing the functional, biotic and physical habitat parameters that are all important to consider when monitoring rivers. A final aim is to evaluate the colonisation trap for its suitability as a tool for citizen science through comparison of invertebrate compositions generated with those of kick-sampling, and through my own experiences of using the traps, suggest improvements that will enhance its viability as a citizen science tool.

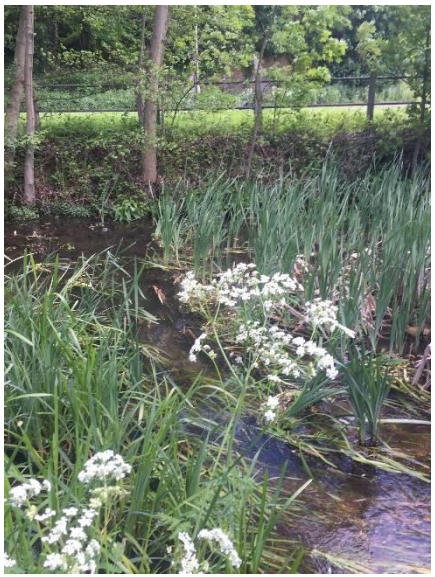
Materials and Methods

Study site

The study area was situated along a small stretch of the River Mimram, Hertfordshire which is a small chalk river approximately 19km in length and is a tributary of the River Lea (The Wild Trout Trust, 2015). There were eight study sites along the upper reaches of the River Mimram around Welwyn: Hoo Farm, Kimpton Mill, Singlers marsh unrestored, Singlers marsh Restored, Digswell Meadow, Tewinbury, Panshanger diversion and Panshanger. The channel dimensions of these sites in 2018 are shown in Table 1 and channel dimensions from 2017 can be found in Zhang (2017). All sites were chosen due to there being corresponding ARMI scores from kick-sampling and MoRPh survey data for the study period, as well as ease of access to the sites and permission from land owners. Four colonisation traps were placed at eight sites along the River Mimram (n=32) to measure invertebrate and microbial decomposition rates and to sample the invertebrate community. All eight sites were sampled in May 2017 (previous study Zhang, 2017), May 2018 and July 2018 to identify annual and seasonal (spring-summer) variation. One site, Hoo Farm was sampled in June 2018 to determine microhabitat variations, with four traps being placed in eight different microhabitats. Microhabitats were identified by site observations and given a name based on their main characteristic visible by sight. The Hoo Farm site was chosen in preference to the other seven sites due to its high microhabitat diversity, as indicated in the MoRPh survey taken in May 2018, site observations and its secluded location, giving a low risk of the traps being disturbed or lost. The microhabitats observed were the Silt edge, exposed channel, water-cress by edge, leaf litter, fast channel in shade, backwater channel edge and *Ranunculus* bed.

Table 1: Channel dimensions of River Mimram sites recorded in May 2018. Sites left to right are moving downstr

				
	Digswell Meadow	Tewinbury	Panshanger diversion	Panshanger
Left bank height(m)	0.45	0.35	0.30	0.20
Right bank height(m)	0.25	0.23	0.30	0.22
Bankfull Width(m)	8.00	11.07	5.50	13.60
Water width(m)	5.90	6.53	4.30	10.20
Water depth(m)	0.45	0.13	0.32	0.21

				
	Hoo Farm	Kimpton Mill	Singlers marsh unrestored	Singlers marsh restored
Left bank height(m)	0.40	0.50	-	0.52
Right bank height(m)	0.46	0.30	0.43	0.50
Bankfull Width(m)	5.63	5.20	5.05	9.50
Water width(m)	4.47	4.70	4.94	8.20
Water depth(m)	0.11	0.10	0.40	0.42

Colonisation traps

Colonisation traps were designed by Murray Thompson and Ian Patmore to measure decomposition rates alongside invertebrate sampling and are made from cuboidal drainpipe tubes, which are separated into two sides by an insert. In one end of the trap (Photo 1) a coarse mesh bag containing a pre-weighed piece of cloth paper is placed and secured inside. In the other end a fine mesh bag containing a pre-weighed piece of cloth paper is placed and secured inside. Each end has a corresponding mesh lid attached to the trap (Zhang, 2017). The use of cloth paper instead of leaves eliminated the need to consider leaf quality and chemistry, which have been found to affect decomposition rates (Tiegs *et al.,* 2007). In preparation for each sample the cloth paper was weighed on a balance to 3 decimal places and the weight recorded alongside the trap number it was placed in and whether it was put in a fine or coarse mesh bag. The traps allowed microorganisms into both sides but prevented invertebrates entering the fine mesh side, an adaptation of a widely recognised method of using mesh bags to measure leaf litter decomposition in rivers (Young *et al.,* 2008).

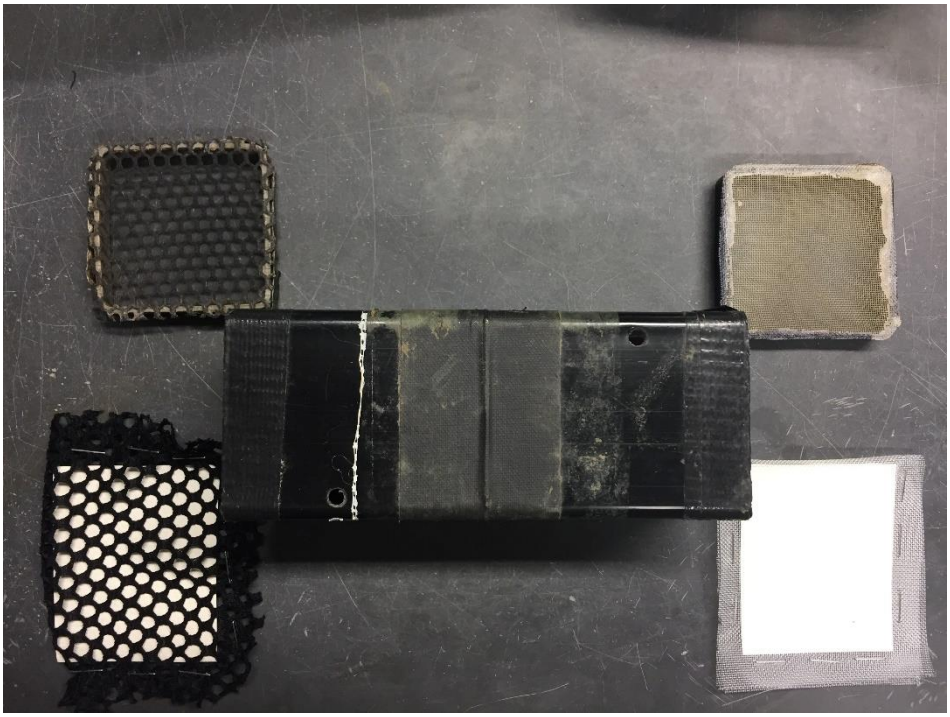


Photo 1: Colonisation trap components including drainpipe tube, coarse and fine mesh lids and coarse and fine mesh bags containing pre-weighed cloth paper.

The traps were secured to the river bed using brackets that fit around the trap and tent pegs were hammered into the river bed. The traps are placed perpendicular to the river flow, allowing water to move in and out and to reduce the flow pressure which may result in traps being swept downstream. The traps were left in the river for two to three weeks to allow for invertebrates and micro-organisms to feed on the paper and colonise the traps. After taking the traps out of the river, the mesh bags were collected to be taken to the lab. There the paper inside the mesh bags was removed, carefully cleaned with water to remove any excess silt on the paper (Gulis *et al.,* 2006), before being placed in individual petri dishes and dried in a gel desiccator oven for four to five days until completely dry (Photo 2). The paper was then weighed to determine its final weight.

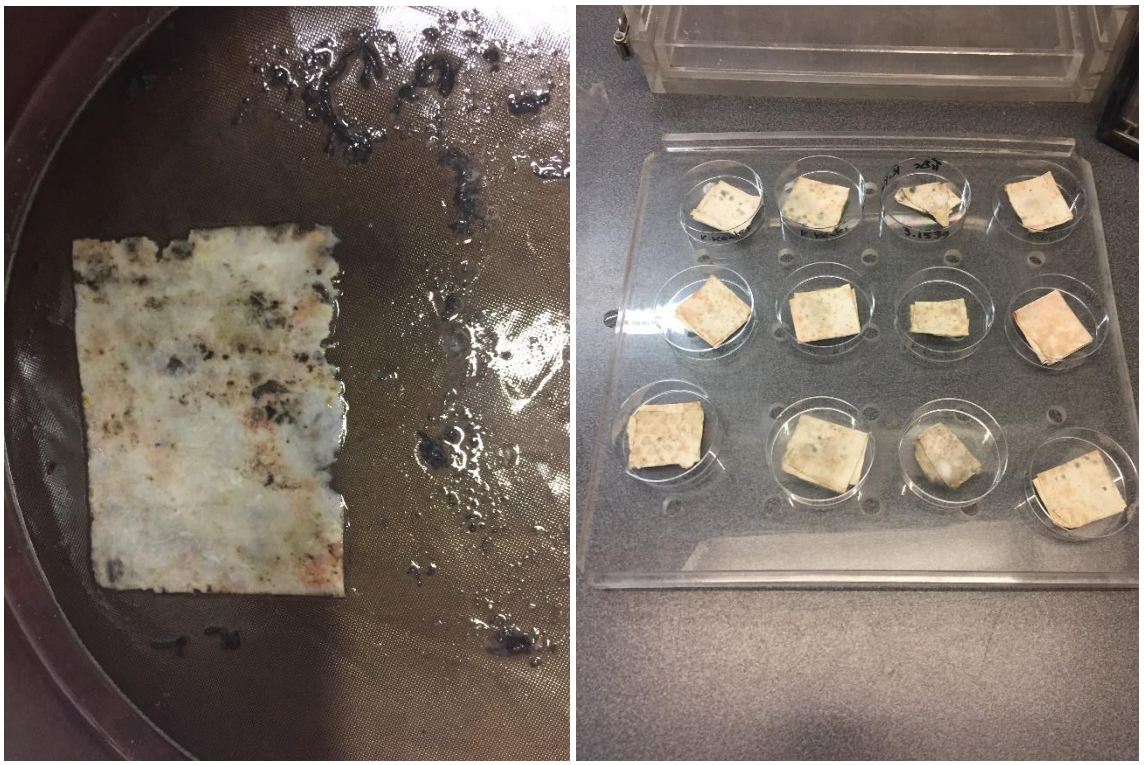


Photo 2: Image on the left-hand side shows paper being cleaned in the lab after it had been taken out of the river. Image on the righthand side shows cleaned paper in petri dishes about to be placed in the gel desiccator to dry.

The change in weight of the cloth paper was calculated for the fine and coarse mesh bags from each trap. This was then divided by the number of days the traps were in the river. This

determined the Rf (microbial decomposition) and Rc (microbial and invertebrate decomposition) per day. The invertebrate decomposition rate per day could then be calculated with the equation: Rc-Rf (Zhang, 2017).

Invertebrate sampling

When the colonisation traps were removed from the river, any invertebrates found inside the coarse mesh end were collected and identified to family level on the riverside, with the abundance of each species being recorded. Invertebrates found on the outside of the traps were not counted. All invertebrate families identified from the colonisation traps and kick-sampling data were classified into one of five functional feeding groups (collector-feeder, collector-gatherer, predator, scraper and shredder) to determine the percentage of each functional feeding group (FFG) at each site (Cummins & Klug, 1979). This is important to determine as shredders play an important role in organic decomposition and their abundance may influence decomposition rates at different sites and spatial variations (Baldy *et al.,* 2002).

Invertebrate abundance, taxon richness and ARMI scores were determined for each colonisation trap. The Riverfly Partnership provided invertebrate kick-sampling data for each site as a comparison to the colonisation traps. ARMI scores were generated by counting the abundance of eight key taxa: cased caddis, caseless caddis, *Ephemeridae, Ephemerellidae, Heptageniidae, Baetidae, Plecoptera* and *Gammaridae*. Each taxa was given a score from 1-4 depending on their abundance as follows1)1-9, 2) 10-99, 3) 100-999 4) 1000+. This score was then summed together to generate an overall ARMI score. The classification of invertebrate families into functional feeding groups was determined using data from West Virginia department of environmental protection (n.d.). This could then be used to determine FFG abundance and from this determine FFG percentage distribution for each site and microhabitat.

MoRPh Survey

The MoRPh survey was carried out at all 8 sites to determine the physical habitat complexity of each site and determine differences between each site and the annual changes by site. The MoRPh survey was designed to enable citizen scientists to monitor local river habitats, surveying river modules for physical characteristics and anthropogenic stressors (Shuker *et al.,*

2017) It is important to determine the physical habitat at each site and in microhabitats to establish whether any variations influence organic decomposition rates and invertebrate composition (Langhans *et al.,* 2008). The MoRPh survey allows for 14 indicies to be determined so that habitat complexity is identified and can be compared across sites. In June the survey was completed for each microhabitat and indices 1-9 were determined to establish if the physical habitat of each microhabitat influenced biotic variables. 2017 data was obtained from online records posted on the MoRPh website (https://modularriversurvey.org/) (Gurnell *et al.,*2016). Only 11 of the indices were used in statistical analysis as indexes 2, 3 and 6 contained categoric data that could not be easily analysed.

Data analysis

Preliminary tests

The statistical program IBM SPSS Statistics 25 was used to carry out the majority of the data analysis. First a One-Sample Kolmogorov-Smirnov Test was carried out to determine if invertebrate decomposition rates, microbe decomposition rate, invertebrate abundance, taxon richness, ARMI scores, functional feeding groups (CF, SC, CG, P and S) and MoRPh indices are normally distributed. Any variables that were not normally distributed were log transformed for better fit of data before being z-transformed for equal weighting of the data set. Some of the MoRPh indices could not be normally distributed and thus non-parametric tests were performed in later analysis where these variables were being used.

Statistical analysis

One-way ANOVA and the post hoc Tukey test were used to determine if there was any significant difference in annual or seasonal variation between the eight sample sites as well as between individual sites. The eight different microhabitats at Hoo Farm were also tested for significant difference using these methods. The non-parametric Kruskal-Wallis Test was used to determine if any of the MoRPh indices were significantly different between 2017 and 2018. Bivariate analysis was performed to determine if any of the MoRPh indices were significantly correlated with the biotic variables such as decomposition rates, invertebrate abundance and richness, ARMI scores and functional feeding groups. This was followed by linear regression to determine what factors if any influenced invertebrate decomposition

rates or any of the other biotic variables and whether this varied over time (annually, seasonally) and between microhabitats.

Canoco (version 5) was used to determine invertebrate taxa distribution across the eight river sites and microhabitats. Correspondence analysis (CA) was performed on seasonal data as an initial detrended correspondence analysis (DCA) produced a gradient length over 4 s.d. units suggesting the unimodal method should be used. Following this a canonical correspondence analysis (CCA) was performed to determine how invertebrate taxa responded to the MoRPh habitat parameters. Principal component analysis (PCA) was performed on annual and microhabitat data due to DCA producing gradient lengths of less than 2 s.d. A PCA was also performed to determine how the river sites responded to changes in FFGs between spring and summer.

Results

Out of the 32 traps placed in the river, only 30 could be sampled in May 2018 due to a trap at Panshanger being lost downstream and another at Tewinbury losing a lid resulting in the loss of the coarse mesh bag. In July only 30 traps could be sampled due to another trap lost downstream at Panshanger and no coarse mesh bag in a trap at Digswell Meadow.

Annual variation

To determine the annual variation between the individual sites and of the River Mimram in total, samples taken in May 2017 were compared to those taken in May 2018. Table 2 shows a summary of the main variances determined from the samples collected in the colonisation traps. There was no significant difference between invertebrate decomposition rates, microbial decomposition rates or ARMI scores in the River Mimram, between 2017 and 2018. There was significantly different invertebrate abundance and taxon richness found between 2017 and 2018 in the River Mimram, with more invertebrate species being found at higher abundance in the colonisation traps in 2018.

Table 2: Comparison of invertebrate and microbial decomposition rates, invertebrate abundance, taxon richness and ARMI score in the River Mimram in 2017 and 2018. Number of samples for each year indicated in brackets. Values are averages with standard deviations. F and P values generated from one-way ANOVA test. Bold values indicate significant difference ($P<0.05$).

	2017 (n=28)	2018 (n=30)	F	P
Invertebrate decomposition rate (g/day)	0.009 ± 0.014	0.005 ± 0.006	1.652	0.204
Microbial decomposition rate (g/day)	0.007 ± 0.004	0.009 ± 0.002	4.015	0.050
Invertebrate abundance	40.643 ± 39.351	101.733 ± 112.801	13.274	**0.001**
Taxon richness	6.071 ± 2.418	8.133 ± 2.675	8.206	**0.006**
ARMI score	3.464 ± 1.774	3.167 ± 1.464	0.037	0.849

Decomposition rates

In 2018 the microbial decomposition rates were on average slightly higher than the invertebrate decomposition rates, which is the reverse of these rates in 2017. The invertebrate decomposition rate at Kimpton Mill in 2017 was significantly higher than that in 2018 and was significantly different to all of the sites in 2017 and 2018 as shown in Figure 1. The other sites showed little variation in invertebrate decomposition between each site and between 2017 and 2018.

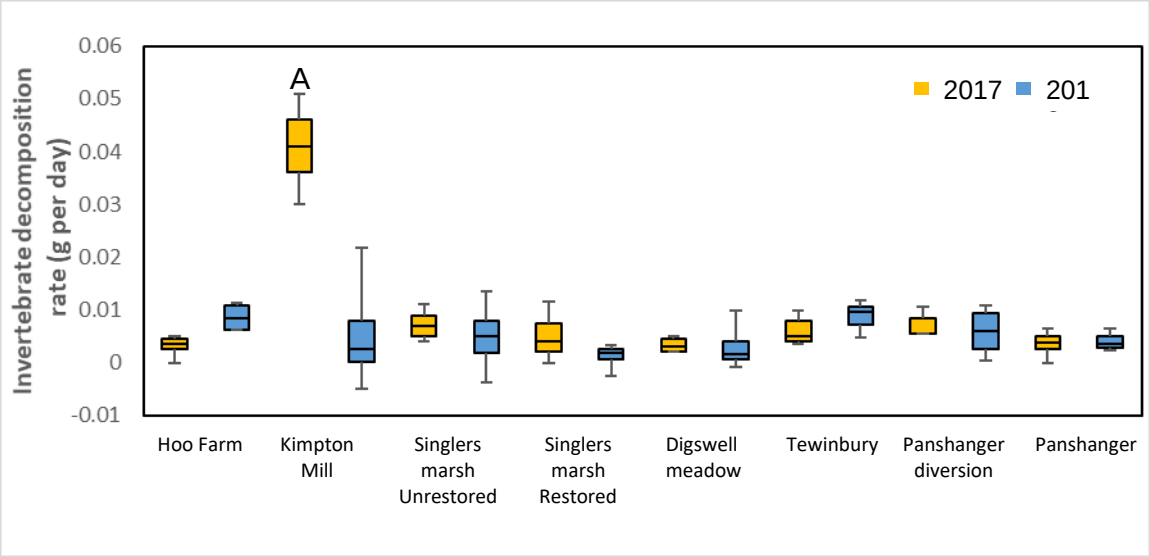


Figure 1: Annual variation in invertebrate decomposition rates for individual sites along the River Mimram. Yellow bars show data from 2017 and blue bars show data from 2018. Error bars show 95% confidence intervals.

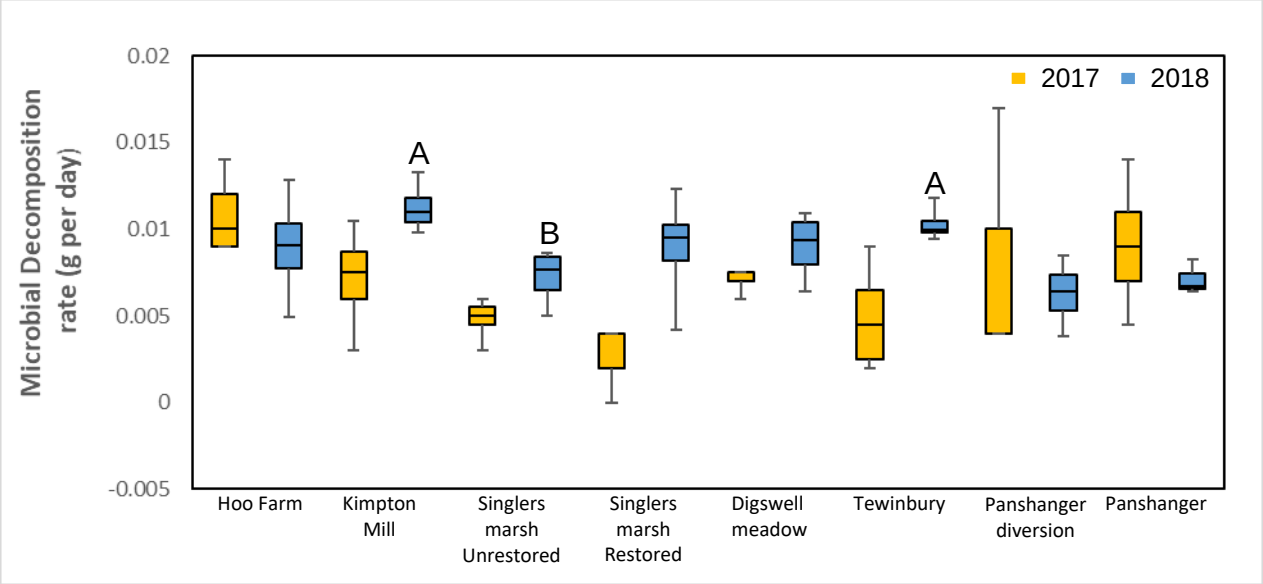


Figure 2: Annual variation in microbial decomposition rates for individual sites along the River Mimram. Yellow bars show data from 2017 and blue bars show data from 2018. Error bars show 95% confidence intervals

There is very little variation between microbial decomposition rates at sites on the River Mimram but some sites in 2018 were significantly different from each other with Singlers Marsh Unrestored significantly lower than Kimpton Mill and Tewinbury as shown in Figure 2.

Invertebrate composition

Invertebrate abundance varied between and within sites, with a high abundance of *Simulium* contributing to higher 2018 abundance, with one trap at Digswell Meadow containing over 600 individuals as well as high numbers found at Tewinbury, Singlers Marsh Restored and Kimpton Mill.Figure 3 shows the distribution in functional feeding groups at each site for 2017 and 2018. There is variation in FFG's across sites with Singlers Marsh unrestored having much higher percentage of scrapers such as snails particularly *P. jenkinsii* found at high abundance at this site. A reduction the percentage of shredders can be seen at many sites from 2017 to 2018 especially at Kimpton Mill and Tewinbury with much lower numbers of *Gammaridae* in 2018. Kick-sampling data from Kimpton Mill 2018 shows large reduction in *Gammaridae* sampled to 2017. For many sites in 2018 the colonisation traps were dominated by collector-filterer species in particular *Simulium*. Sites in 2017 also show higher percentage

of collector-gatherers and predators than 2018 with much higher abundances of *Ephemerellidae* and *Turbellaria* respectively in the traps.

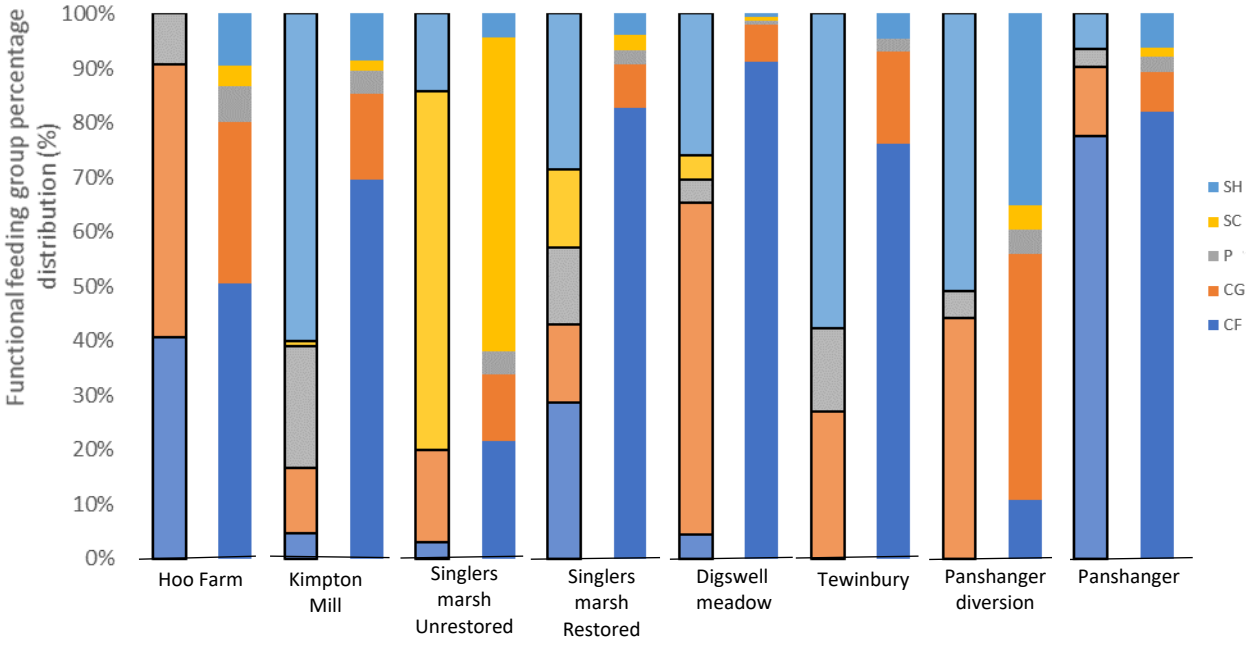


Figure 3: Annual variation in invertebrate functional feeding group percentage for each site. For each site 2017 and 2018 data is paired together with 2017 data to the left with columns outlined in black and 2018 data is to the right and columns are not outlined. Five functional feeding groups are SH=shredder, SC=scraper, P=predator, CG=collector-gatherer and CF=collector-filterer.

Physical habitat complexity

The 11 MoRPh indices statistically analysed were compared for annual variation (Table2) with the number of bed material types and channel physical habitat complexity being significantly different between 2017 and 2018. There were higher numbers of bed material types seen at Panshanger diversion, Tewinbury and Kimpton Mill in 2018 to 2017, and slightly higher channel physical habitat complexity seen at most sites in 2018.

Table 2: The comparison of MoRPh indices from 2017 to 2018. Averages are shown with standard deviations. F and P values determined by Kruskal-Wallis test with bold values indicating significance (P<0.05).

	2017 (n=8)	2018 (n=8)	F	P
Number of flow types	1.125 ± 0.354	1.125 ± 0.354	0.000	1.000
Number of bed material types	1.625 ± 0.518	3.25 ± 1.165	7.656	**0.006**
Average bed material particle size (phi units)	0.75 ± 2.629	1.311 ± 3.007	0.045	0.833
Extent of bed siltation	0.063 ± 0.177	0.000	1.000	0.317
Channel physical habitat complexity	1.375 ± 0.700	1.834 ± 0.308	5.128	**0.024**
Number of aquatic vegetation morphotypes	2.125 ± 0.991	3.000 ± 1.309	1.731	0.188
Riparian physical habitat complexity	1.143 ± 0.319	0.973 ± 0.387	0.471	0.493
Riparian vegetation complexity	4.688 ± 0.821	6.375 ± 1.778	3.202	0.074
Degree of human pressure imposed by land cover on the bank tops	1.156 ± 1.420	0.313 ± 0.884	3.506	0.061
Channel reinforcement	0.375 ± 0.694	0.833 ± 1.541	0.077	0.782
Extent of non-native invasive plants	0.094 ± 0.186	0.500 ± 0.720	0.799	0.371

Interactions between biotic and physical habitat parameters

None of the MoRPh indices or the other biotic variables were found to significantly influence invertebrate decomposition in 2018. Pearson correlation of 2018 environmental and biotic variables showed channel physical habitat complexity was positively correlated to scraper abundance, riparian physical habitat complexity was positively correlated to collector-filterer abundance and riparian vegetation complexity as positively correlated to microbial decomposition, collector-gatherer abundance and predator abundance. Environmental variables correlated negatively with biotic variables in 2017 (Table3).

Table3: Pearson correlation for environmental and biotic variables for 2017 and 2018. Vales show strength of correlation with bold values indicating significance (P<0.05).

	Average bed material particle size		Channel physical habitat complexity		Number of aquatic vegetation morphotypes		Riparian physical habitat complexity		Riparian vegetation complexity	
	2017	2018	2017	2018	2017	2018	2017	2018	2017	2018
Invertebrate decomposition rate (g/day)	-0.277	-0.481	-0.275	0.327	0.286	-0.332	-0.487	0.479	-0.407	0.172
Microbial decomposition rate (g/day)	**-0.680**	-0.261	0.174	-0.489	-0.003	-0.488	-0.285	0.469	-0.550	**0.645**
Invertebrate abundance	**-0.764**	0.061	-0.028	-0.577	0.178	-0.392	-0.571	0.562	**-0.758**	-0.047
Taxon richness	**-0.939**	0.119	-0.060	-0.499	0.376	0.345	-0.331	0.058	**-0.642**	0.196
ARMI score	-0.586	-0.374	0.508	-0.184	-0.302	0.525	-0.096	0.539	-0.570	-0.056
CF%	0.002	-0.576	**0.710**	-0.221	**-0.651**	-0.105	0.405	**0.842**	0.314	-0.427
CG%	-0.180	0.473	-0.421	-0.454	0.480	0.098	-0.141	-0.409	-0.349	**0.734**
SH%	0.154	-0.014	-0.174	0.474	0.047	0.397	0.105	-0.380	-0.348	-0.234
P%	-0.220	-0.008	-0.298	-0.073	0.389	-0.602	-0.038	0.027	-0.133	**0.658**
SC%	0.425	0.388	-0.092	**0.630**	-0.034	-0.602	-0.282	-0.572	0.522	-0.003

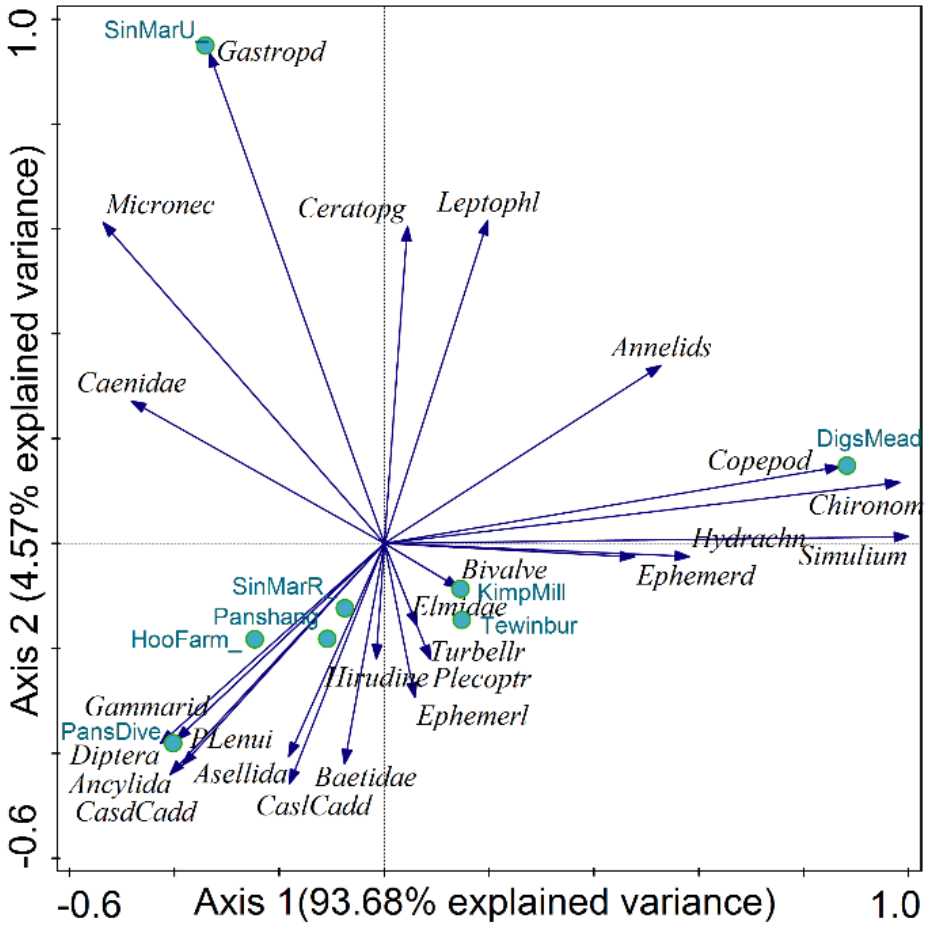


Figure 4: PCA ordination plot showing species associations to 2018 River Mimram sites. Blue arrows indicate species association and light blue circles indicate River Mimram 2018 sites. Axis 1 and 2 show 98.25% explained variance.

Figure 4 indicates interactions between 2018 sites and invertebrate species. Most 2018 sites are grouped together indicating similar species composition. Singlers Marsh Unrestored has a strong association with the second axis and has a distinct species community including *Gastropods*, *Micronecta* and *Caenidae*. Digwell Meadow has a strong association with the first axis and also has a distinct invertebrate compostion including *Simulium, Copepods, Chironomids, Hydrachnidae* and *Ephermeridae*.

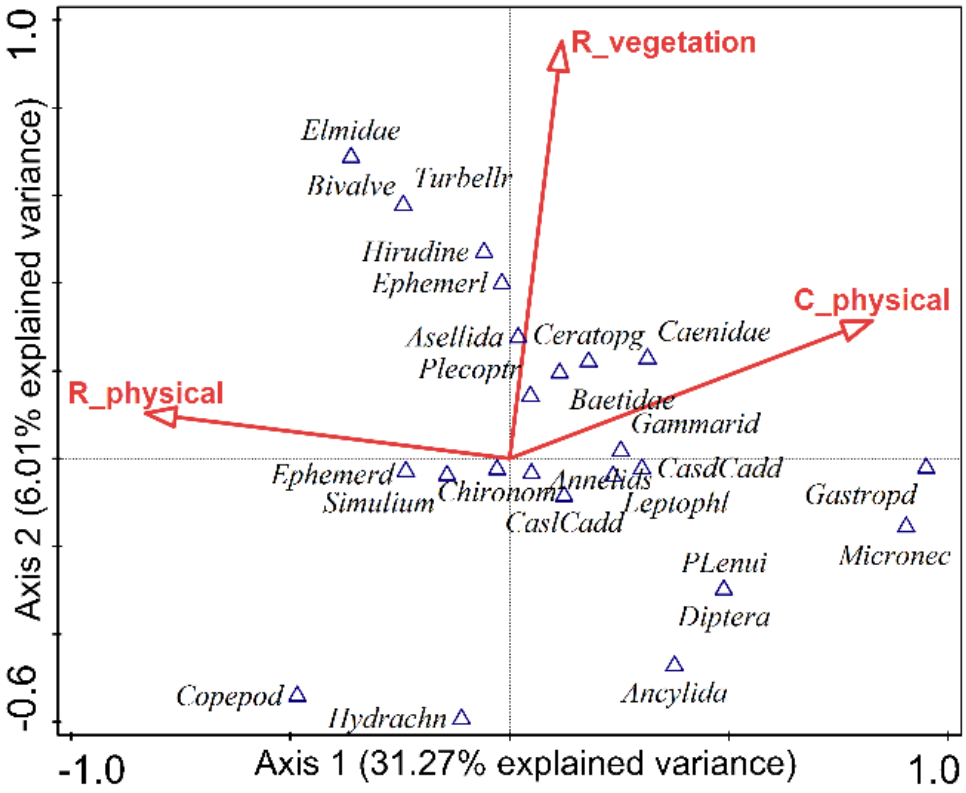


Figure 5: CCA constrained plot shows associations between the physical habitat parameters R physical (riparian physical habitat complexity), R vegetation (riparian vegetation complexity) and C physical (channel physical habitat complexity) to invertebrate taxa for River Mimram 2018 data. Red arrows indicate environment parameters and blue triangles indicate invertebrate taxa. Axis 1 and 2 shows 37.28% explained variance.

The CCA plot (figure 5) shows R physical associated with axis 1 with *Simulium*, *Ephermeridae* and *Chironomids*. R vegetation is strongly associated with axis 2 as are the taxa Asellidae, Ephermerellidae and *Hirudinea*.

Comparison of techniques for sampling invertebrate community

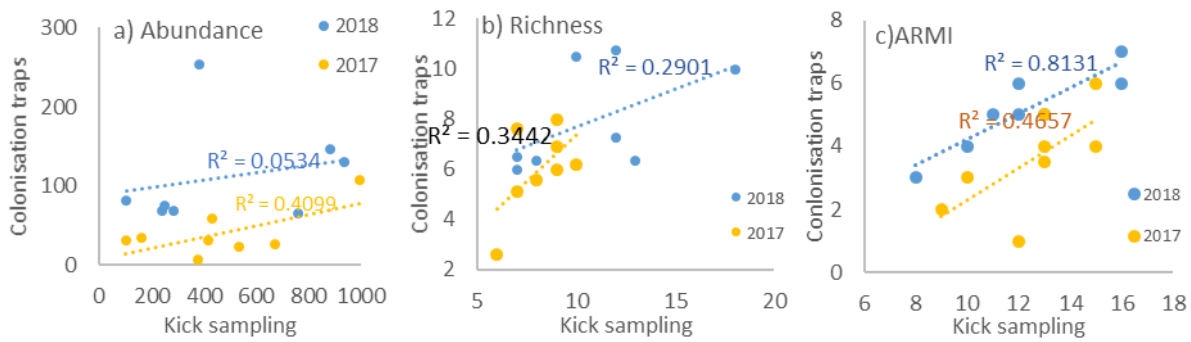


Figure 6: Comparison in a) invertebrate abundance, b) taxon richness and c) AMRI scores between colonisation traps and kick-sampling methodology, and annual variation. 2017 data show in yellow and 2018 data shown in blue with linear trendlines shown and R^2 values are shown for each.

Invertebrate sampling through colonisation traps is linearly related to kick-sampling in both years with abundance and ARMI score regression lines having similar slope gradients in 2017 and 2018 (Figure 6). A strong positive relationship was seen between methods for ARMI scores. Invertebrate richness differed in 2018 to 2017 with more species being recorded for both methods. In all three indexes kick-sampling produces higher results.

Seasonal variation

Seasonal variation in decomposition rates and invertebrate composition and abundance was determined by collection of samples in May and July to compare Spring with Summer. A one-way ANOVA test indicated significant difference in invertebrate decomposition rates, microbial decomposition rates and invertebrate abundance between Spring and Summer as shown in Table 4. Decomposition rates for both invertebrates and microbial were higher in the summer but invertebrate abundance was lower. In July one trap at Kimpton Mill and two at Tewinbury had no cloth paper left in the coarse mesh bag when collected and so the invertebrate decomposition rates for these sites may be higher than the results indicate.

Table 4: Comparison of invertebrate and microbial decomposition rates, invertebrate abundance, taxon richness and ARMI score between spring and summer. Values are averages with standard deviations. F and P values generated from one-way ANOVA test. Bold values indicate significant difference ($P<0.05$).

	Spring (n=30)	Summer (n=30)	F	P
Invertebrate decomposition rate (g/day)	0.005 ± 0.006	0.010 ± 0.005	12.928	**0.001**
Microbial decomposition rate (g/day)	0.009 ± 0.002	0.010 ± 0.003	4.559	**0.037**
Invertebrate abundance	101.733 ± 112.801	45.781 ± 24.820	6.896	**0.011**
Taxon richness	8.133 ± 2.675	7.656 ± 2.377	0.089	0.767
ARMI score	3.167 ± 1.464	3.871 ± 2.045	1.893	0.174

Decomposition rates

Invertebrate decomposition had little variation across sites in both spring and summer as shown in Figure 7. The three most up-stream sites had similar decomposition rates between spring and summer. The five most downstream sites showed higher invertebrate decomposition in summer than in spring although this was not significantly different. There was significant difference between summer decomposition rates at Tewinbury and spring decomposition rates at Singlers Marsh Restored and Digswell Meadow.

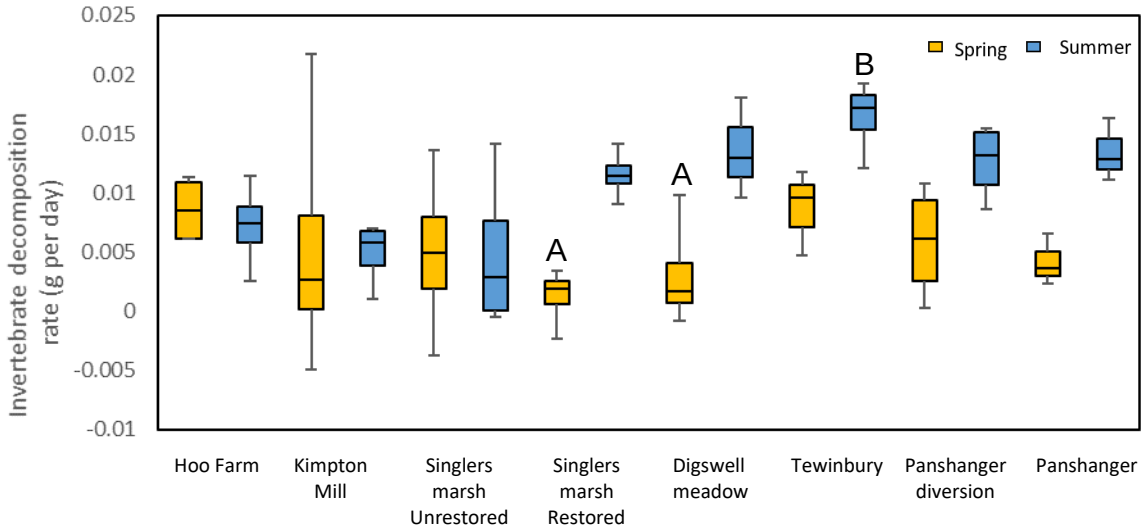


Figure 7: Seasonal variation in invertebrate decomposition rates for individual sites along the River Mimram. Yellow bars show data from Spring and blue bars show data from Summer. Error bars show 95% confidence intervals.

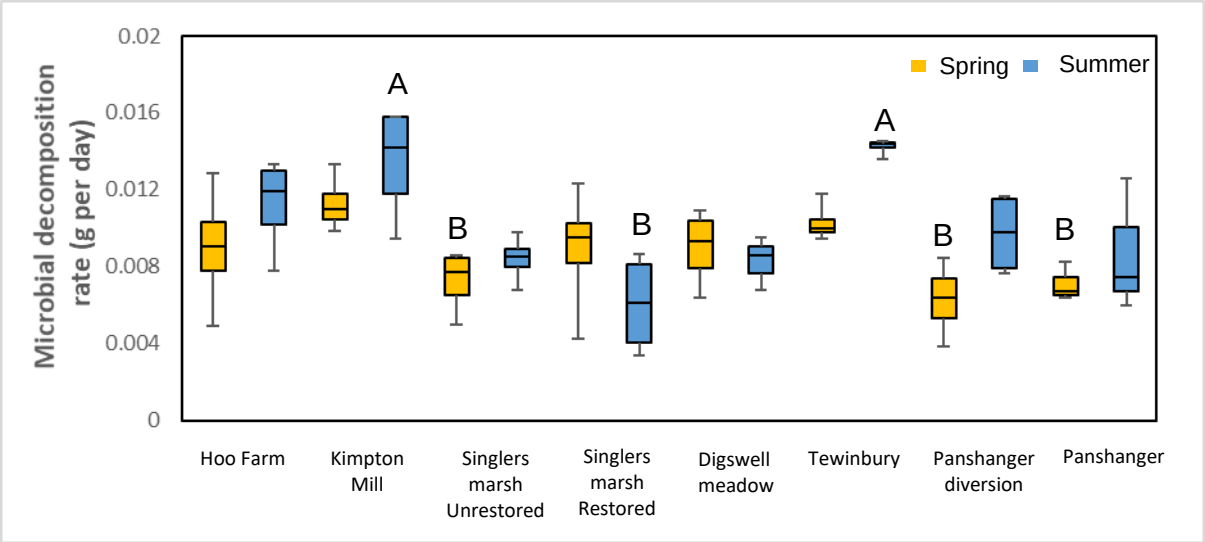


Figure 8: Seasonal variation in microbial decomposition rates for individual sites along the River Mimram. Yellow bars show data from spring and blue bars show data from summer. Error bars show 95% confidence intervals.

Microbial decomposition rates were significantly different between some sites in the summer with Singlers Marsh Restored having lower a decomposition rate than Kimpton Mill and Tewinbury as shown in figure 8. Both these sites had significantly higher microbial decomposition rates in the summer than three sites in the spring.

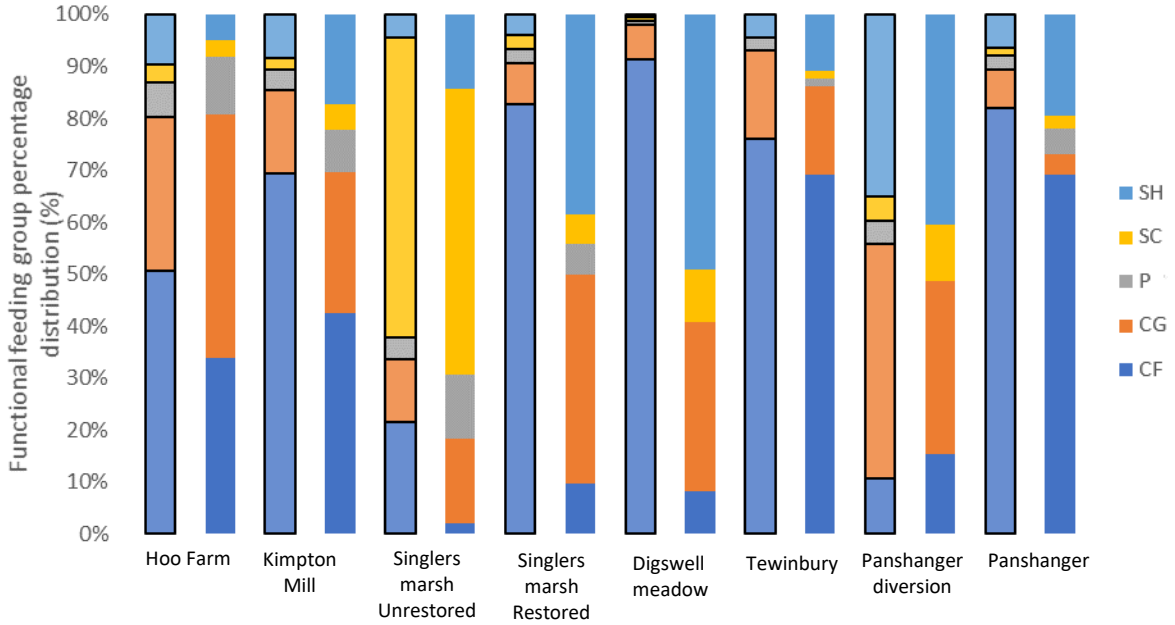


Figure 9: Seasonal variation in invertebrate functional feeding group percentage for each site. For each site spring and summer data is paired together with May data to the left with

columns outlined in black and July data is to the right and columns are not outlined. Five functional feeding groups are SH=shredder, SC=scraper, P=predator, CG=collector-gatherer and CF=collector-filterers.

Most sites had similar functional feeding group distributions between spring and summer (figure 9). However Digswell Meadow and Singlers Marsh Restored went from very high CF % in the spring to below 10% in the summer, with higher distributions of shredders such as *Gammaridae* and *Leuctridae* and CGs seen in the summer. There were significantly more shredders found at sites in the summer (ANOVA, F=6.424, p=0.014) and more collector=gatherers (ANOVA, F=6.229, p=0.015) than the spring. Very low abundance of *Simulium* were collected from the colonisation traps in the summer compared to the spring at most sites with much lower abundances seen.

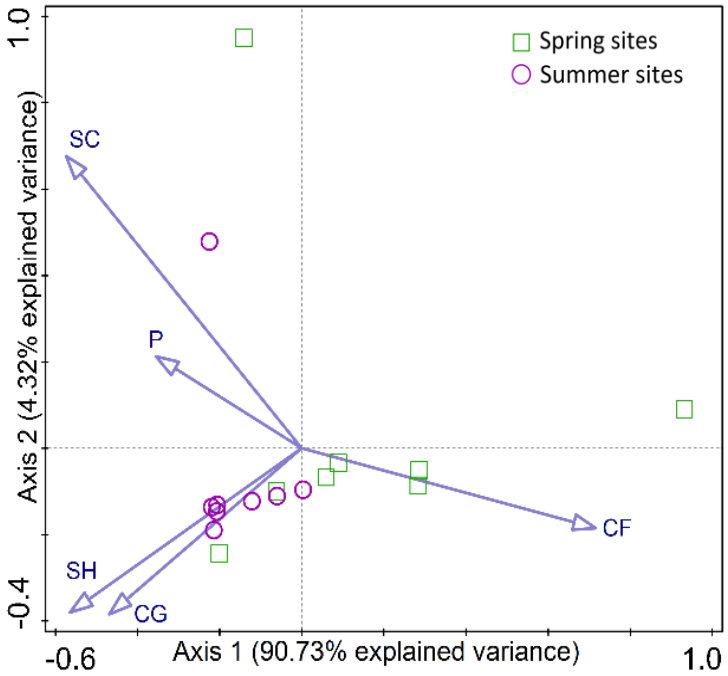


Figure 10: CCA ordination plot indicating associations between five different FFGs and sites in spring and summer. Blue arrows indicate the five FFGs (SH=shredder, SC=scraper, P=predator, CG=collector-gatherer and CF=collector-filterers). Green squares show spring sites and purple circles show summer sites. Axis 1 and 2 show 95.05% explained variance.

Most summer sites are highly associated with Shredders and collector-gathers apart from Singlers Marsh Unrestored which is associated with scrapers for both the spring and summer. Spring sites are more varied with most having high association with collector-filterers.

Microhabitat variation

Table 5 shows the overall variation in invertebrate and microbial decomposition rates, species abundance, taxon richness and ARMI scores across eight river microhabitats. There was no significant difference in invertebrate decomposition rates across microhabitats (figure 11). There was significant difference in microbial decomposition rates between microhabitats with further post hoc tests determined that silt edge and exposed channel microhabitats were significantly different with the silt edge having a significantly lower microbial decomposition rate (figure 12). Species abundance across microhabitats was significantly different. The post hoc test Tukey determined that species abundance in the fast channel in shade microhabitat was significantly higher than the water-cress by edge, leaf litter, backwater and channel edge habitats (figure 13). There was no significant difference in invertebrate taxon richness or ARMI scores across microhabitats.

Table 5: Comparison of invertebrate and microbial decomposition rates, invertebrate abundance, taxon richness and ARMI score between microhabitats. Values are averages with standard deviations. F and P values generated from one-way ANOVA test. Bold values indicate significant difference ($P<0.05$).

	Microhabitats (n=32)	F	P
Invertebrate decomposition rate (g/day)	0.009 ± 0.005	1.941	0.107
Microbial decomposition rate (g/day)	0.011 ± 0.003	3.261	**0.014**
Invertebrate abundance	39.000 ± 23.654	6.318	**<0.001**
Taxon richness	7.488 ± 1.545	0.910	0.516
ARMI score	2.563 ± 0.982	1.007	0.451

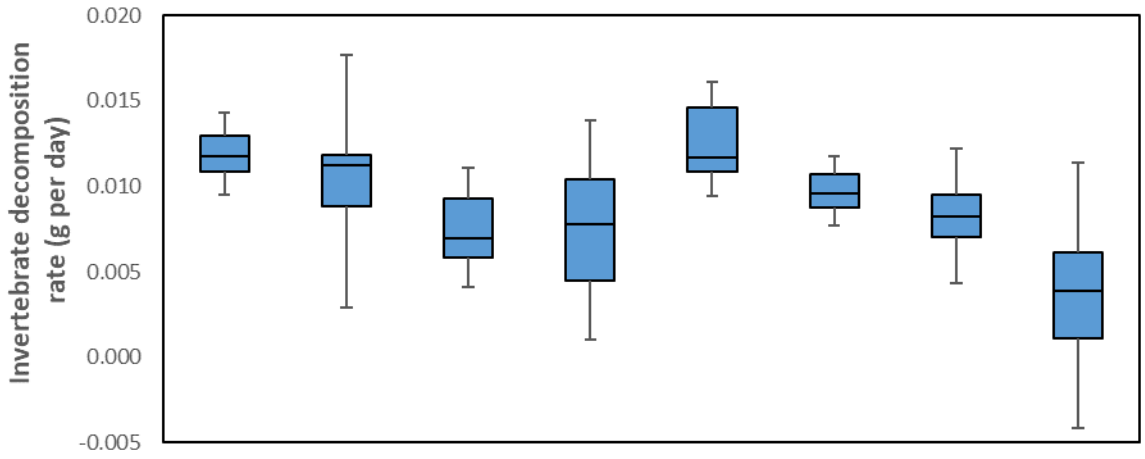


Figure 11: Variation in invertebrate decomposition across eight microhabitats at a site on the River Mimram. Error bars show 95% confidence intervals.

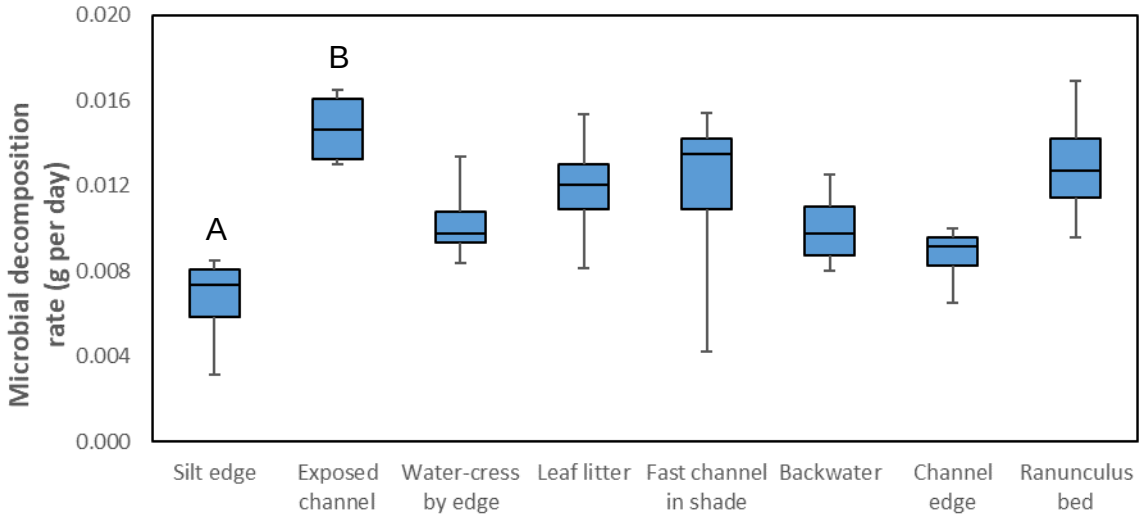


Figure 12: Variation in microbial decomposition across eight microhabitats at a site on the River Mimram. Letters (A, B) show microhabitats that are statistically significantly different (ANOVA, Turkey; $P<0.05$). Error bars show 95% confidence intervals.

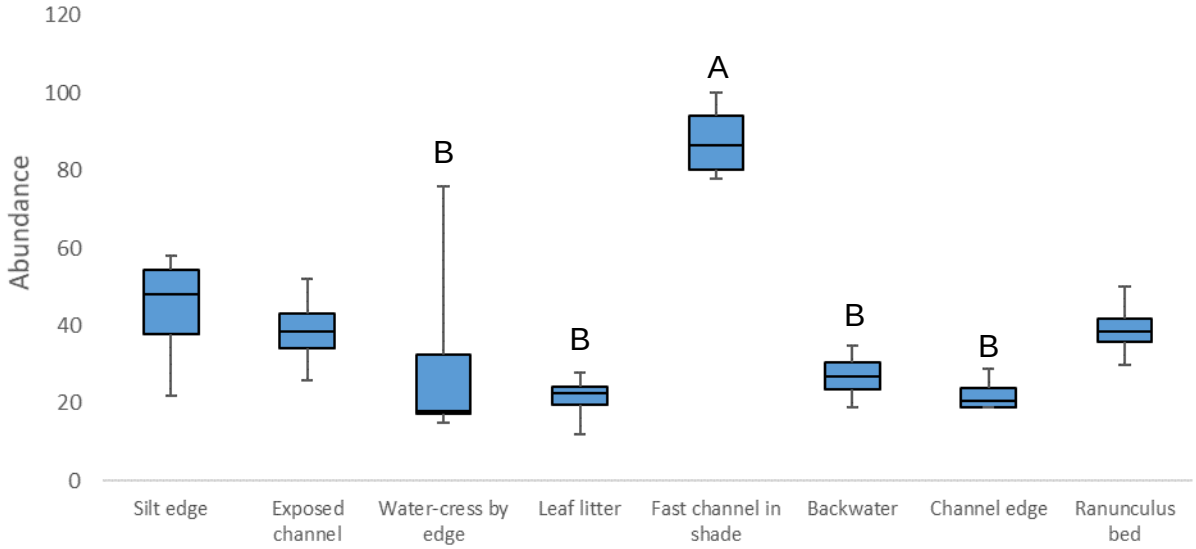


Figure 13: Variation in species abundance across eight microhabitats. Letters (A, B) show microhabitats that are statistically significantly different (ANOVA, Turkey; P<0.05). Error bars show 95% confidence intervals.

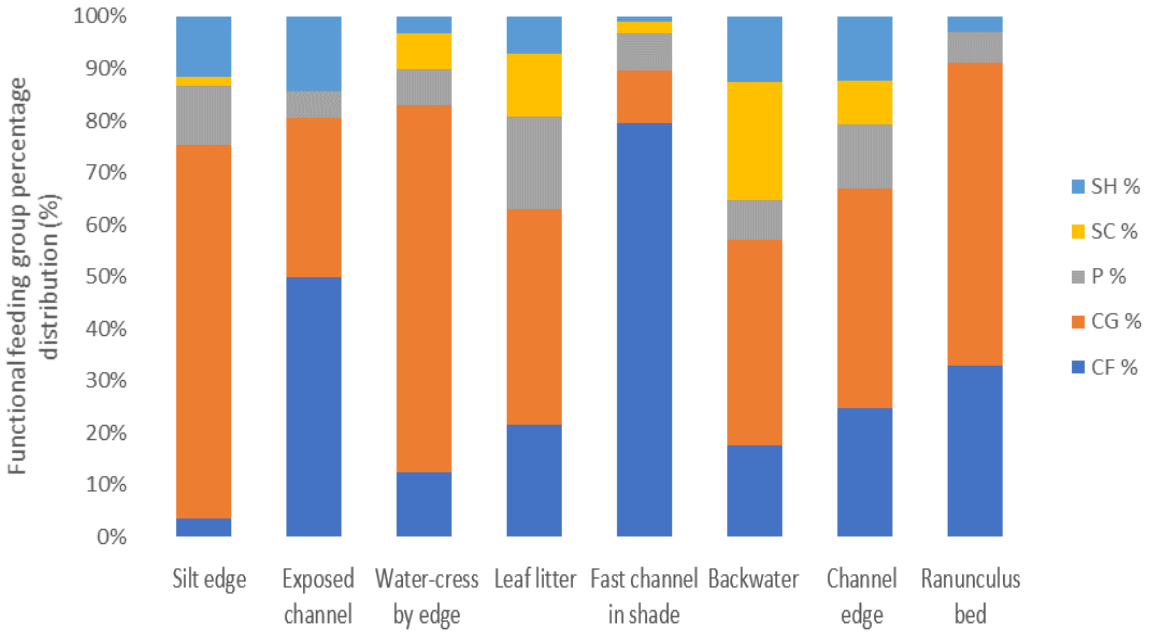


Figure 14: Invertebrate functional feeding group variation at eight microhabitats. Stacked columns show percentage of each functional feeding group (SH=shredder, SC=scraper, P=predator, CG=collector-gatherer and CF=collector-filterer.

An Independent-Samples Kruskal-Wallis test determined that CF, CG and SC varied significantly across microhabitats (P<0.05) with the fast channel in shade having significantly

higher CF to the silt edge, and lower CG to the silt edge and water cress by edge. The backwater had significantly higher SC to the exposed channel, *Ranunculus* bed and the silt edge (figure 14).

The PCA ordination plot shown in figure 15 shows the interactions between invertebrate taxa and the different microhabitats. The silt edge microhabitat has a particular species composition differing to the other microhabitats including *Asellidae*, *Glossiphonid*, *Annelids* and *Ostracods*. The fast channel microhabitat has a distinct species community the caseless caddis species *Polycentropodidae* and *Rhyacopilidae*, the cased caddis *Leptoceridae* and *Simulium*. The majority of the microhabitats have more similar invertebrate communities especially the leaf litter, channel edge and backwater microhabitats which are associated with *Planorbid* snails, *Caenidae* and *Goerids* among many other species.

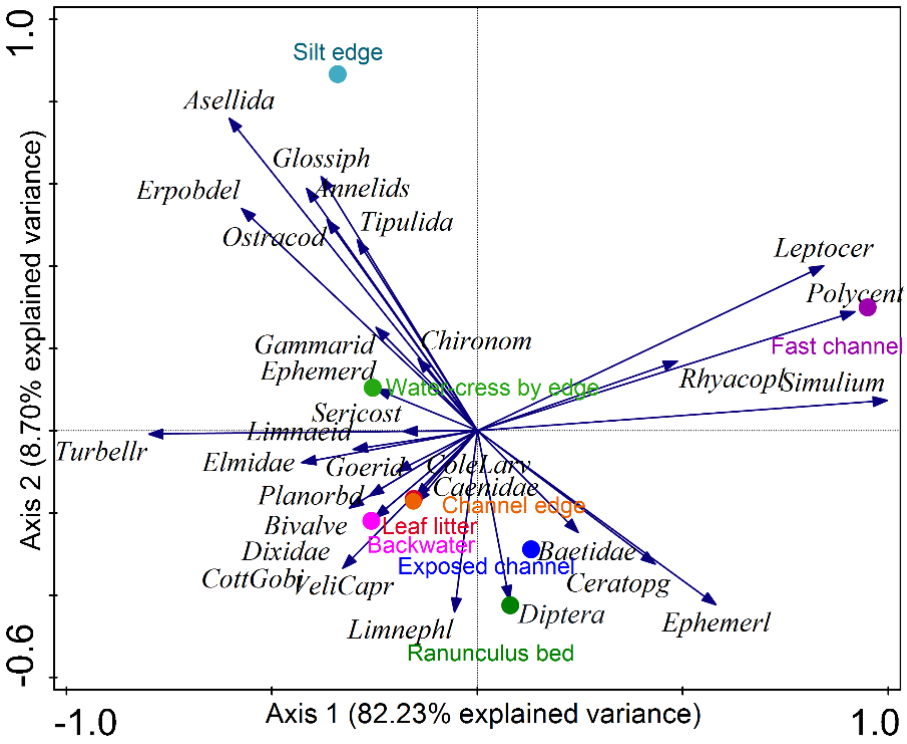


Figure 15: PCA ordination plot indicating species associations with microhabitats. Taxa interactions are indicated by arrows. Microhabitat plots are indicated by coloured circles. Axis 1 and 2 shows 90.93% explained variance.

Linear regression analysis determined that microhabitats invertebrate decomposition rates were found to be influenced by invertebrate abundance (p=0.06) and taxon richness (p=0.049). With an increase in invertebrate abundance and a decrease in taxon richness

linked to increased decomposition rates. Invertebrate decomposition rates were also influenced by the number of bed material particle size (p=0.032) and channel physical habitat complexity (p=0.033). Increased decomposition rates are influenced by increased number of bed material particle size and decreased channel physical habitat complexity. Microbial decomposition rates and taxon richness were also significantly influenced by channel physical habitat complexity.

Discussion

Annual variation

There was no annual variation in invertebrate or microbial decomposition rates on the River Mimram. Unlike the 2018 data the 2017 data showed slightly higher invertebrate decomposition rates than microbial decomposition rates, which is what would be expected from a small temperate river (Gonçalves Jr *et al.,* 2006). Zhang (2017) found that the high invertebrate decomposition rates at Kimpton Mill in 2017 was probably influenced by high shredder abundance in the form of *Gammaridae*. The lack of high shredder abundance at Kimpton Mill in 2018 may therefore explain the lower decomposition rate recorded, although invertebrate decomposition rates in 2018 were not found to be influenced significantly by any biotic parameters, including shredder abundance. Microbial decomposition rates were higher at some sites in 2018 compared to 2017 but this did not translate into overall significantly increased microbial decomposition in the River Mimram in 2018.
Invertebrate abundance was significantly higher in 2018 compared to 2017 and this is probably due to very high numbers of *Simulium* found at several sites in 2018, including Digswell Meadow where over 600 were counted as well as over 100 counted at Singlers Marsh Restored and Tewinbury. This was also reflected in the functional feeding group % distribution, as collector-filterers made-up over 80% of the invertebrate composition at these sites. It is unclear why the *Simulium* were found in such high numbers in the traps, as they feed on fine particulate organic matter (McCullough *et al.,* 1979). However, large numbers were also often found on the outside of the traps as well, including empty larval cases, indicating that they might use the traps as substratum to attach to (Mathuriau & Chauvet, 2002). There was also a large reduction in shredder species distribution in 2018 compared to

2017 at many sites, specifically *Gammaridae*. This was also seen in the kick-sampling data with many sites having lower *Gammaridae* abundance, suggesting that the colonisation traps can show the same trends as the kick-sampling for some species. There were also much lower numbers of Turbellaria found in the traps in 2018 leading to a reduced percentage of predators. For both 2017 and 2018 Singlers Marsh Unrestored stands out, with a completely different FFG% make-up from the rest of the sites, with very high numbers of *P. jenkinsii* a non-native mud snail that is often found in disturbed habitats, in shallow water and areas with high siltation (Van Damme, 2013).

Physical parameters of a river site, such as habitat heterogeneity, can influence biotic functions such as organic decomposition rates (Frainer *et al.,* 2017). There were strong negative correlations found between average bed material particle size and the biotic parameters' microbial decomposition, invertebrate abundance and taxon richness. Changes in bed material size have been shown to influence microbial communities and their spatial distribution (Swan & Palmer, 2001). There was a strong positive correlation between riparian vegetation complexity and microbial decomposition, possibly due to higher vegetation complexity providing more habitats for micro-organisms to colonise, increasing microbial abundances in that area. The 2017 data largely showed negative correlations between environmental parameters and biotic parameters but the opposite was seen in the 2018 data, with some parameters having positive correlations.

The positive correlations between the physical habitat parameters and specific FFG's for the 2018 data was shown in more detail in the CCA ordination plot, where collector-filterers *Simulium* and *Ephermeridae* were associated with riparian physical habitat complexity. Taniguchi & Tokeshi (2004) found that *Simulium* were found in low abundances in high habitat complexity as they prefer high-flow, unobstructed surfaces associated with low habitat complexity. Gastropods and were associated with channel physical habitat complexity. *Turbellaria, Hirudinea*, *Ephemerellidae* and *Asellidae* were associated with riparian vegetation complexity, which had a strong association with the second canonical axis.

Seasonal variability

Invertebrate and microbial decomposition rates both significantly increased from spring to summer. This is most probably linked to increased temperatures, which have been seen to increase invertebrate decomposition rates (Anderson & Sedell, 1979) and microbial decomposition rates probably enabling increased biological activity and metabolic rate (Pascoal & Cássio, 2004)

Large changes occur within the river ecosystem between spring and summer, such has increased temperatures and increased vegetation (Champion & Tanner, 2000). Invertebrate abundance significantly reduced from spring to summer and this was mainly due to the large populations of *Simulium* sampled in spring not being present in the summer. This was also seen in the change in FFG % in the summer, where there was a large reduction in CF% across the sites, with the exception of Tewinbury and Panshanger. Several sites had higher invertebrate decomposition in the summer than the spring, although not statistically different. These sites were those further downstream while the upstream sites remained more constant. There was an increase in SH% at the same downstream sites, which may explain higher decomposition rates seen in the summer at these particular sites (Graça, 2001). In the summer Tewinbury had high invertebrate and microbial decomposition rates but had lower SH% than the other downstream sites which would not have been expected. However, there was very little of the cloth paper remaining in the traps at this site, with two having no paper remaining. The increased decomposition rates were probably due to water-cress which grew around the traps after they were placed in the river, which would have increased habitat heterogeneity (Frainer *et al.,* 2017). The high decomposition rates suggest higher presence of shredders at Tewinbury but, with no paper left in some of the traps, shredders, such as *Gammaridae*, may have been lower in number than expected because there was no food left for them. The three most up-stream sites had little increase in SH%, which may also indicate why they did not have higher invertebrate decomposition in the summer. Overall canonical analysis indicated that in the spring collector-filterers were the most abundant functional feeding group, while in the summer shredders and collector-gatherers made up a large percent of the invertebrate community.

Microhabitat variation

There was no significant difference in invertebrate decomposition rates between microhabitat sites although this has been shown by Kominoski & Rosemond (2012). They determined that there were hot spots in leaf litter decomposition in a headwater stream, with litter patches formed in the middle of pools having higher decomposition rates than those formed in riffles or on the edge or alcove of the pools. There was a significant difference in microbial decomposition between the silt edge and the exposed channel, with the silt edge having lower microbial decomposition. This may be due to microbial decomposition rates being influenced by channel physical habitat complexity. The MoRPh survey indicated that the silt edge habitat had a low channel physical habitat complexity compared to the exposed channel. Invertebrate abundance was significantly higher in the fast channel in shade microhabitat, which had high invertebrate decomposition rates in relation to the other microhabitats but was not significantly different. There was no significant difference in taxon richness or ARMI scores between microhabitats. Increased invertebrate decomposition rates were shown to be influenced by increased invertebrate abundance and the number of bed material particle sizes, as well as decreased taxon richness and channel physical habitat complexity. Studies have shown that different microhabitats can contain different invertebrate communities and different species abundance and diversity (Lamouroux *et al.,* 2004). This was shown in the significantly different FFGs distribution between microhabitats with the slower flowing microhabitats, such as the silt edge, water-cress by edge and backwater having different compositions to the fast channel in shade and the exposed channel. The difference in flow regimes between microhabitats can cause different invertebrate communities to be formed as some species prefer fast flowing regime while other prefer slower (Bunn & Arthington, 2002). This was indicated with the fast channel having a high presence of collector-filterers such as *Simulium*, while the backwater had a high occurrence of scrapers such a Planorbids. The fast channel and silt edge also had different invertebrate compositions in relation to each other and the other microhabitats, indicating the importance of sampling different microhabitats using the colonisation traps to further understand factors affecting decomposition rates and invertebrate communities.

Comparing colonisation traps with kick-sampling

When comparing the invertebrate data from the colonisation traps to kick-sampling, linear relationships are shown. Increases in invertebrate abundance, richness and ARMI scores

when kick-sampling also shown in the colonisation traps. Kick-sampling obtains much larger samples for each parameter especially for invertebrate abundance, which is 10x bigger in kick-sampling than colonisation traps, and this impacts the ARMI scores generated from the traps, as higher scores are gained through large abundances of mayflies and *Gammaridae*. However the ARMI scores show the strongest linear relationship between kick-sampling and traps and this was shown in both 2017 and 2018 data. There was no significant difference in taxon richness between 2017 and 2018 suggesting that similar species may be using the traps. However, the kick-samples generate higher taxon richness indicating that there are some species that do not use the traps and are not represented when surveying the invertebrate community with the traps, so it is important to use another method alongside the traps. Although the kick-sampling data for these sites showed higher species diversity than the colonisation traps, the ARMI survey only requires identification of eight species, most of which are flies (Huddart *et al.,* 2016). This means that most volunteers will only survey these particular species, which, although is sufficient for monitoring changes in water quality, may not give a true representation of the invertebrate community. This is the case for Singlers Marsh Unrestored, which has a high abundance of *Gastropods* that would not necessarily be recorded by ARMI volunteers. However, their presence in such high abundances may indicate important ecological information about that site (Lewin, 2006). By leaving the colonisation traps in the river for weeks at a time it may allow different species, which are not picked up by short kick-sampling durations, to be identified, including rare species (Mykrä *et al.,* 2006) For volunteers who wish to further their knowledge of their local river and its ecology, the ability, on occasion to look at more invertebrate species and to successfully identify them may increase their engagement with the topic and encourage them to contribute more to data collection (Tweddle *et al.,* 2012).

Evaluating the use of colonisation traps for citizen science

Using the traps

The colonisation traps are an only recently used way of sampling organic decomposition rates alongside invertebrates. The design of the trap allows re-use, unlike single use mesh bags, and they are hard wearing, unaffected by being in the river for long periods of time. The lids generally prevent the mesh bags from being lost downstream and the traps provide a standardised size for invertebrates to colonise and feed off the paper inside. Making up

numerous traps is time-consuming but not difficult and requires little training whilst following simple instructions. Considerable time could be saved if there was an easier way to attach the lids to the traps as they are currently secured using tape. This would also save time when taking the lids off at the river site to retrieve the mesh bags and sample the invertebrates. Ideally a balance capable of weighing to three decimal places is required to weigh each paper before and after it is placed into the river, as the loss in weight can be quite small and variations may not be identified at 2 decimal places. This might maker it harder for volunteers to do but pre-weighed paper could be sent out and samples sent back to a lab to be weighed, as is done with other variables such as eDNA analysis (Biggs *et al.,* 2015). Once the traps are made up it is easy to place them in the river correctly and could be correctly completed by citizen science volunteers following simple instructions. Taking them out of the river is also straightforward, with the main factors to be aware of being: taking out the traps without losing the invertebrates inside, placing the mesh bags in named sample bags. The identification of invertebrates down to families would require a volunteer with previous knowledge or training but organisations running citizen science projects such as ZSL or The Riverfly Partnership do offer training in identification for volunteers performing kick-samples and similar training could be given to those who would be sampling the colonisation traps (Fore *et al.,* 2008). The data for the River Mimram showed that species richness was not very high so it is probable that volunteers would only need to identify a few more species than those used for the ARMI scores, provided that the river they are sampling does not have higher species richness than the river Mimram.

After the paper has been in the river it can be fiddly to clean off any excess silt without losing bits of paper and this can be harder if the paper is more fragmented. The paper can be air dried anywhere but will take several days and so may not be suitable for some citizen scientists. It may be better for samples to be sent to a lab or to allocate responsibility for cleaning, drying and weighing the paper to specific individuals or a co-ordinator (Roy *et al.,* 2012). This may also help with data quality as it reduces human error caused by different techniques being used by different people (Dickinson *et al.,* 2010). The weight change data could easily be recorded on a datasheet to determine decomposition rates per day, and invertebrate data could also be uploaded to a datasheet for a national data bank to be created (Crall *et al.,* 2010).

Data generated from the traps

The colonisation traps allow a different parameter to be measured, one that is not currently being used for citizen science. The use of a new parameter, such as decomposition rates, could increase the environmental data collected for a river and widen the number of factors monitored (Conrad & Hilchey, 2011). Measuring decomposition rates allows for a different way of thinking about the river ecosystem, not just the biology or chemistry of the river (Lecerf *et al.,* 2006). There have been several studies indicating that organic decomposition rates are important parts of the river system function which can influence many areas and are therefore important to monitor (Powell, 2014). Some studies suggest decomposition rates are more reliable than invertebrate or other biological indicators (Pozo *et al.,* 2011). There is still so much unknown about decomposition rates and how they are influenced that it is important that baseline data can be generated for different rivers and microhabitats over time in order that rates generated through volunteer monitoring in the future can be compared against a baseline and can be seen as an indicator of good or bad river health (Dudgeon, 2010; Tank *et al.,* 2010). The colonisation traps are a recent introduction and, with only a small amount of data obtained from them to date, it is important to do further studies to determine how well they work and how the data they generate can be used alongside other bio-monitors. The concept of decomposition rates may be harder for the general public to understand and may not hold the excitement behind kick-sampling or pond netting but, by including the invertebrate sampling with the decomposition, it may allow more people to start to be interested in the topic and want to learn more (Bonney *et al.,* 2009).

Future studies

Future studies leading on from this project could look at similar microhabitats at different sites to determine whether there is any similarity within specific microhabitats and could enable baseline data for particular microhabitats on the River Mimram to be obtained. It would also be beneficial to compare the microhabitat invertebrate communities sampled in the colonisation traps to kick-sampling or other methodologies to find out if there is a linear relationship and to identify what taxa the colonisation traps might have missed.

It would be important to measure environmental parameters such as temperature, nutrient level, dissolved oxygen, flow rates and how variations in these parameters might influence invertebrate and microbial decomposition rates.

In relation to the citizen science aspects of using the traps, a future study with the traps deployed by volunteers is required to determine how the volunteers respond to using the traps, what they think of deploying them alongside kick-sampling and to try to identify any problems with the methodology and trap design that might arise.

Conclusion

Although there was no significant variation in annual invertebrate or microbial decomposition rates variation was seen between some sites and there is a high probability that the large reduction in invertebrate decomposition rate at Kimpton Mill from 2017 to 2018 is due to the large reduction in shredder abundance. Physical habitat parameters were negatively correlated with biotic parameters in 2017 and positively correlated in 2018. Seasonal variations in invertebrate and microbial decomposition rates were seen with high rates in the summer probably linked to warmer temperatures which can increase biological activity in the river. Although not found to be significantly influencing invertebrate decomposition rates, higher distribution of shredders at downstream sites in the summer could have caused increased decomposition rates.

Microhabitat variation was seen in microbial decomposition rates with the silt edge having lower decomposition than the exposed channel. There was also significant differences in FFG distributions in several of the microhabitats, with flow regime variation possibly being a factor in differing invertebrate composition between some of the microhabitats. Other factors were discovered to be influencing invertebrate and microbial decomposition rates, including taxon richness and channel physical habitat complexity.

Linear relationships were seen between colonisation traps and kick-sampling with invertebrate abundance and ARMI scores showing similar relationships in 2017 and 2018. Overall the colonisation traps were successful in helping to determine decomposition rates across River Mimram sites and between microhabitats. There are a few negatives with the traps, such as using tape to keep the lids on makes them time-consuming to put together and creates a lot of waste at the end. However, with further understanding of decomposition rates and their role as indicators of river health becoming more developed, new monitoring techniques such as the colonisation traps could allow for citizen scientists to further their knowledge in river ecosystems and keep them engaged in their local environment.

Auto-critique

I chose to do this project as I was interested in looking at citizen science techniques used for monitoring of aquatic systems. I had never thought about leaf litter decomposition rates as something to monitor for river health and so I was intrigued to learn more about this part of the river ecosystem and how it could be linked to citizen science through the use of colonisation traps.

A strength of the study was the high number of different biotic and physical habitat parameters that were surveyed and analysed, this allowed for more in depth look into decomposition rates and what my influence then. Being able to use previous year's data to compare against also gave another aspect to the project.

More MoRPh survey data was needed for increased statistical analysis, as sample number was low compared to the number of variables and most data was not normally distributed.

Improvements I would have made if I could do it again would be to take the MoRPh survey in July so that changes in spring and summer physical habitat. On observation when returing to the sites in July there were changes in vegetation abundance that could have influenced invertebrate composition and decomposition rates.

References

Abell, R., Thieme, M.L., Revenga, C., Bryer, M., Kottelat, M., Bogutskaya, N. Coad, B., Mandrak., N., Contreras Balderas, S., Bussing, W., Stiassny, M.L.J., Skelton, P., Allen, G.R., Unmack, P., Naseka, A., Ng, R., Sindorf, N., Robertson, J., Armijo, E., Higgins, J.V., Heibel, T.J., Wikramanayake, E., Olsen, D., López, H.L., Reis, R.E., Lundberg, J.G., Sabaj Pérez, M.H. & Petry, P. (2008) Freshwater ecoregions of the world: A new map of biogeographic units for freshwater biodiversity conservation. *BioScience.* Vol 58(5): 403-414

Anderson, N.H. & Sedell, J.R. (1979). Detritus processing by macroinvertebrates in stream ecosystems. *Annual Review of Entomology.* Vol 24: 351-377

Aristi, I., Díez, J.R., Larrañaga, A., Navarro-Ortega, A., Barceló, D. & Elosegi, A. (2012). Assessing the effects of multiple stressors on the functioning of Mediterranean rivers using poplar wood breakdown. *Science of The Total Environment.* Vol 440: 272-279

Baldy, V., Chauvet, E., Charcosset, J-Y. & Gessner, M.O. (2002). Microbial dynamics associated with leaves decomposing in the mainstem and floodplain pond of a large river. *AME.* Vol 28: 25-36

Battarbee, R.W., Shilland, E.M., Kernan, M., Monteith, D.T. & Curtis, C.J. (2014). Recovery of acidified surface waters from acidification in the United Kingdom after twenty years of chemical and biological monitoring (1988–2008). *Ecological Indicators.* Vol 37(Part B): 267-273

Beavan, L., Sadler, J. & Pinder, C. (2001). The invertebrate fauna of a physically modified urban river. *Hydrobiologia.* Vol 445: 97-108

Bennett, C., Owen, R., Birk, S., Buffagni, A., Erba, S., Nicolas, M., Murray-Bligh, J., Ofenbock, G., Pardo, I., Van de Bund, W., Wagner, F. & Wasson, J-G. (2011). Bringing European river quality into line: an exercise to intercalibrate macro-invertebrate classification methods. *Hydrobiologia.* Vol 667: 31-48

Biggs, J., Ewald, N., Valentini, A., Gaboriaud, C., Dejean, T., Griffiths, R.A., Foster, J., Wilkinson, J.W., Arnell, A., Brotherton, P., Williams, P. & Dunn, F. (2015). Using eDNA to develop a national citizen science-based monitoring programme for the great crested newt (*Triturus cristatus*). *Biological Conservation.* Vol 183: 19-28

Black, J.M. (2009). River Otter monitoring by citizen science volunteers in Northern California: Social groups and litter size. *Northwestern Naturalist.* Vol 90(2): 130-135

Bonney, R., Cooper, C.B., Dickinson, J., Kelling, S., Phillips, T., Rosenburg, K.V. & Shirk, J. (2009). Citizen Science: A Developing Tool for Expanding Science Knowledge and Scientific Literacy. *BioScience.* Vol 59(11): 977-984

Bonney, R., Shirk., J.L., Phillips, T.B., Wiggins, A., Ballard, H.L., Miller-Rushing, A.J. & Parrish, J.K. (2014). Next Steps for Citizen Science. *Science*. Vol 343(6178): 1436-1437

Buytaert, W., Zulkafli, Z., Grainger, S., Acosta, L., Alemie, T.C., Bastiaensen, J., De Bièvre, B., Bhusal, J., Clark, J., Dewulf, A., Foggin, M., Hannah, D.M., Hergarten, C., Isaeva, A., Karpouzoglou, T., Pandeya, B., Paudel, D., Sharma, K., Steenhuis, T., Tilahun, S., Van Hecken, G. & Zhumanova, M. (2014). Citizen science in hydrology and water resources: opportunities for knowledge generation, ecosystem service management, and sustainable development. *Frontiers in Earth Science.* Vol 2(26): 1-21

Bunn, S.E. & Arthington, A.H. (2002). Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management.* Vol 30(4): 492-507

Champion, P.D. & Tanner, C.C. (2000). Seasonality of macrophytes and interaction with flow in a New Zealand lowland stream. *Hydrobiologia.* Vol 441(1): 1-12

Crall, A.W., Newman, G.J., Jarnevich, C.S., Stohlgren, T.J., Waller, D.M. & Graham, J. (2010). Improving and integrating data on invasive species collected by citizen scientists. *Biological Invasions.* Vol 12(10): 3419-3428

Clarke, R.T., Wright, J.F. & Furse, M.T. (2003). RIVPACS models for predicting the expected macroinvertebrate fauna and assessing the ecological quality of rivers. *Ecological Modelling.* Vol 160(3): 219-233

Conrad, C.C. & Hilchey, K.G. (2011). A review of citizen science and community-based environmental monitoring: issues and opportunities. *Environmental Monitoring and Assessment.* Vol 176(1-4): 273-291

Costa, S.S. & Melo, A.S. (2008). Beta diversity in stream macroinvertebrate assemblages: among-site and among-microhabitat components. *Hydrobiologia.* Vol 598(1): 131-138

Cummins, K.W. & Klug, M.J. (1979). Feeding Ecology of stream invertebrates. *Annual Review of Ecology and Systems.* Vol 10: 147-72

Dickinson, J.L., Shirk, J., Bonter, D., Bonney, R., Crain, R.L., Martin, J., Phillips, T. & Purcell, K. (2012). The current state of citizen science as a tool for ecological research and public engagement. *Frontiers in Ecology and the Environment.* Vol 10(6): 291-297

Dickinson, J.L., Zuckerberg, B. & Bonter, D.N. (2010). Citizen science as an Ecological Research Tool: Challenges and benefits. *Annual Review of Ecology, Evolution, and Systematics.* Vol 41: 149-172

Dudgeon, D. (2010). Prospects for sustaining freshwater biodiversity in the 21st century: linking ecosystem structure and function. *Current Opinion in Environmental Sustainability.* Vol 2(5-6): 422-430

Dunkley, R.A. (2018). Monitoring ecological change in UK woodlands and rivers: An exploration of the relational geographies of citizen science. *Transactions of the Institute of British Geographers.* Vol 0(0): 1-16

Everard, M. (2008). Selection of taxa as indicators of river and freshwater wetland quality in the UK. *Aquatic Conservation: Marine and Freshwater Ecosystems.* Vol 18: 1052-1061

Feio, M.J., Alves, T., Boavida, M., Medeiros, A. & Graça, M.A.S. (2010). Functional indicators of stream health: a river-basin approach. *Freshwater Biology.* Vol 55(5): 1050-1065

Ferreira, V. & Canhoto, C. (2014). Effect of experimental and seasonal warming on litter decomposition in a temperate stream. *Aquatic Sciences.* Vol 76(2): 155-163

Frainer, A., Polvi, L.E., Jansson, R. & McKie, B.G. (2017). Enhanced ecosystem functioning following stream restoration: The roles of habitat heterogeneity and invertebrate species traits. *Journal of Applied Ecology.* Vol 55(1): 377-385

Fore, L.S., Paulsen, K. & O'Laughlin, K. (2008). Assessing the performance of volunteers in monitoring streams. *Freshwater Biology.* Vol 46(1): 109-123

Griffiths, M. (2002). The European Water Framework Directive: an approach to integrated river basin management. *E-WAter*, 1-15

Graça, M.A.S. (2001). The Role of Invertebrates on Leaf Litter Decomposition in Streams – a Review. *International Review of Hydrobiology.* Vol 86(4-5): 383-393

Gonçalves Jr, J.F., Graça, M.A.S. & Callisto, M. (2006). Leaf-litter breakdown in 3 streams in temperate, Mediterranean, and tropical Cerrado climates. *Freshwater Science.* Vol 25(2): 344-355

Gulis, V., Ferreira, V. & Graça, M.A.S. (2006). Stimulation of leaf litter decomposition and associated fungi and invertebrates by moderate eutrophication: implications for stream assessment. *Freshwater Biology.* Vol 51: 1655-1669

Gurnell, A., Shuker, L. & Wharton, G. (2016). Modular River Survey. (WWW.) Downloaded on 26th August 2018 from https://modularriversurvey.org/

Huddart, J.E.A., Thompson, M.S.A., Woodward, G. & Brooks, S.J. (2016). Citizen science: from detecting pollution to evaluating ecological restoration. *Wiley Interdisciplinary Reviews: Water*. Vol 3(3): 287-300

Iliopoulou-Georgudaki, J., Kantzaris, V., Katharios, P., Kaspiris, P., Georgiadis, Th. & Montesantou, B. (2003). An application of different bioindicators for assessing water quality: a case study in the rivers Alfeios and Pineios (Peloponnisos, Greece). *Ecological Indicators.* Vol 2(4): 345-360

Kallis, G., Butler, D. (2001). The EU water framework directive: measures and implications. *Water Policy.* Vol 3(2): 125-142

Kobayashi, S. & Kagaya, T. (2005). Hot spots of leaf breakdown within a headwater stream reach: comparing breakdown rates among litter patch types with different macroinvertebrate assemblages. *Freshwater Biology.* Vol 50(6): 921-929

Kominoski, J.S. & Rosemond, A.D. (2012). Conservation from the bottom up: forecasting effects of global change on dynamics of organic matter and management needs for river networks. *Freshwater Science.* Vol 31(1): 51-68

Lamouroux, N., Dolédec, S. & Gayraud, S. (2004). Biological traits of stream macroinvertebrate communities: effects of microhabitat, reach, and basin filters. *Freshwater Science.* Vol 23(3): 449-466

Langhans, S.D., Tiegs, S.D., Gessner, M.O. & Tockner, K. (2008). Leaf-decomposition heterogeneity across a riverine floodplain mosaic. Aquatic Sciences. Vol 70(3): 337-346

Lecerf, A., Usseglio-Polatera, P., Charcosset, J-Y., Lambrigot, D., Bracht, B. & Chauvet, E. (2006). Assessment of functional integrity of eutrophic streams using litter breakdown and benthic macroinvertebrates. *Archiv für Hydrobiologie.* Vol 165(1): 105-126

Lewin, I. (2006). The *Gastropod* communities in the lowland rivers of agricultural areas - their biodiversity and bioindicative value in the Ciechanowska upland, central Poland. *Malacologia.* Vol 49(1): 7-23

Leaf Pack Network. (2018). Linking the Leaf Pack Experiment to Stream Ecology. (WWW.) Downloaded 24th August 2018 from https://www.leafpacknetwork.org/learn/studies/

Lye, G.C., Osborne, J.L., Park, K.J. & Goulson, D. (2012). Using citizen science to monitor Bombus populations in the UK: nesting ecology and relative abundance in the urban environment. *Journal of Insect Conservation.* Vol 16(5): 697-707

Magurran, A.E., Baillie, S.R., Buckland, S.T., Dick, J.McP., Elson, D.A., Scott, E.M., Smith, R.I., Somerfield, P.J. & Watt, A.D. (2010). Long-term datasets in biodiversity research and monitoring: assessing change in ecological communities through time. *Trends in Ecology and Evolution.* Vol 25(10): 574-582

Malmqvist, B. (2002). Aquatic invertebrates in riverine landscapes. *Freshwater Biology.* Vol 47(4): 679-694

Mathuriau, C. & Chauvet, E. (2002). Breakdown of leaf litter in a neotropical stream. *Freshwater Science.* Vol 21(3): 384-396

McCullough, D.A., Minshall, G.W. & Cushing, C.E. (1979). Bioenergetics of lotic filter-feeding insects *Simulium spp*. (Diptera) and *Hydropsyche occidentalis* (Trichoptera) and their function in controlling organic transport in streams. *Ecology.* Vol 60(3): 585-596

McKinley, D.C., Miller-Rushing, A.J., Ballard, H.L., Bonney, R., Brown, H., Cook-Patton, S.C., Evans, D.M., French, R.A., Parrish, J.K., Phillips, T.B., Ryan, S.F., Shanley, L.A., Shirk, J.L., Stepenuck, K.F., Weltzin, J.F., Wiggins, A., Boyle, O.D., Briggs, R.D., Chapin III, S.F., Hewitt, D.A., Preuss, P.W. & Soukup, M.A. (2017). Citizen science can improve conservation science, natural resource management, and environmental protection. *Biological Conservation.* Vol 208: 15-28

Mykrä, H., Ruokonen, T. & Muotka, T. (2006). The effect of sample duration on the efficiency of kick-sampling in two streams with contrasting substratum heterogeneity. *Internationale Vereinigung für theoretische und angewandte Limnologie: Verhandlungen*. Vol 29(3): 1351-1355

Navel, S., Mermillod-Blondin, F., Montuelle, B., Chauvet, E., Simon, L., Piscart, C. & Marmonier, P. (2010). Interactions between fauna and sediment control the breakdown of plant matter in river sediments. *Freshwater Biology.* Vol 55(4): 753-766

Newman, G., Wiggins, A., Crall, A., Graham E., Newman, S. & Crowston, K. (2012). The future of citizen science: emerging technologies and shifting paradigms. *Frontiers in Ecology and the Environment.* Vol 10(6): 298-304

Niyogi, D.K., Harding, J.S. & Simon, K.S. (2013). Organic matter breakdown as a measure of stream health in New Zealand streams affected by acid mine drainage. *Ecological Indicators.* Vol 24: 510-517

Overdevest, C., Huyck Orr, C. & Stepenuck, K. (2004). Volunteer stream monitoring and local participation in Natural Resource issues. *Human Ecology Review.* Vol 11(2): 177-185

Pascoal, C. & Cássio, F. (2004). Contribution of Fungi and Bacteria to Leaf Litter Decomposition in a Polluted River. *Applied and Environmental Microbiology.* Vol 70(9): 5266-5273

Pascoal, C., Pinho, M., Cássio, F. & Gomes, P. (2003). Assessing structural and functional ecosystem condition using leaf breakdown: studies on a polluted river. *Freshwater Biology.* Vol 48(11): 2033-2044

Powell, T. (2014) The design, development and testing of a novel freshwater biomonitoring tool. Imperial College London, unpublished thesis.

Power, M.E., Parker, M.S. & Dietrich, W.E. (2008). Seasonal reassembly of a river food web: floods, droughts, and impacts of fish. *Ecological Monographs.* Vol 78(2): 263-282

Pozo, J., Casas, J., Menéndez, M., Mollá, S., Arostegui, I., Basaguren, A., Casado, C., Descals, E., García-Avilés, J., González, J.M., Larrañaga, A., López, E., Lusi, M., Moya, O., Pérez, J., Riera, T., Roblas, N. & Salinas, M.J. (2011). Leaf-litter decomposition in headwater streams: a comparison of the process among four climatic regions. *Freshwater Science.* Vol 30(4): 935-950

Rose, N.L., Turner, S.D., Goldsmith, B., Gosling, L. & Davidson, T.A. (2016). Quality control in public participation assessments of water quality: the OPAL Water Survey. *BMC Ecology.* Vol 16(Suppl 1) S14: 23-43

Roy, H.E., Pocock, M.J.O., Preston, C.D., Roy, D.B., Savage, J., Tweddle, J.C. & Robinson, L.D. (2012) Understanding citizen science and environmental monitoring: final report on behalf of UK Environmental Observation Framework. Wallingford, NERC/Centre for Ecology & Hydrology, 173pp.

Shuker, L.J., Gurnell, A.N., Wharton, G., Gurnell, D.J. England, J., Leeming, B.F.F. & Beach, E. (2017). MoRPh: a citizen science tool for monitoring and appraising physical habitat changes in rivers. *Water and Environment Journal.* Vol 31(3): 418-424

Silvertown, J. (2009). A new dawn for citizen science. *Trends in Ecology and Evolution.* Vol 24(9): 467-471

Steenweg, R., Hebblewhite, M., Kays, R., Ahumada, J., Fisher, J.T., Burton, C., Townsend, S.E., Carbone, C., Rowcliffe, J.M., Whittington, J., Brodie, J., Royle, J.A., Switalski, A., Clevenger, A.P., Heim, N. & Rich, L.N. (2016). Scaling-up camera traps: monitoring the planet's biodiversity with networks of remote sensors. *Frontiers in Ecology and the Environment*. Vol 15(1): 26-34

Storey, R.G. & Wright-Stow, A. (2017). Community-based monitoring of New Zealand stream macroinvertebrates: agreement between volunteer and professional assessments and performance of volunteer indices. *New Zealand Journal of Marine and Freshwater Research.* Vol 51(1): 60-77

Storey, R.G., Wright-Stow, A., Kin, E., Davies-Colley, R.J. & Stott, R. (2016). Volunteer stream monitoring: Do the data quality and monitoring experience support increased community involvement in freshwater decision making? *Ecology and Society.* Vol 21(4): 32

Swan, C.M. & Palmer, M.A. (2001). What drives small-scale spatial patterns in lotic meiofauna communities? *Freshwater Biology.* Vol 44(1): 109-121

Syberg, K., Hansen, S.F., Christensen, T.B. & Khan, F.R. (2018). 'Risk Perception of Plastic Pollution: Importance of Stakeholder Involvement and Citizen Science', in Wagner M., Lambert S. (eds) *Freshwater Microplastics. The Handbook of Environmental Chemistry, vol 58*. Cham: Springer, 203-221

Taniguchi, H. & Tokeshi, M. (2004). Effects of habitat complexity on benthic assemblages in a variable environment. *Freshwater Biology.* Vol 49(9): 1164-1178

Tank, J.L., Rosi-Marshall, E.J., Griffiths, N.A., Entrekin, S.A. & Stephen, M.L. (2010). A review of allochthonous organic matter dynamics and metabolism in streams. *Freshwater Science.* Vol 29(1): 118-146

Tiegs, S.D., Langhans, S.D., Tockner, K. & Gessner, M.O. (2007). Cotton strips as a leaf surrogate to measure decomposition in river floodplain habitats. *Freshwater Science.* Vol 26(11): 70-77

The Riverfly Partnership. (n.d.). Anglers' Riverfly Monitoring Initiative. The Riverfly Partnership. (WWW). Downloaded on 24th August 2018 from: http://riverflies.org/rp-riverfly-monitoring-initiative

The Wild Trout Trust. (2015). River Mimram, Tewin, near Welwyn Garden City, Hertfordshire. An Advisory Visit by the Wild Trout Trust October 2015.

Tulloch, A.I.T., Possingham, H.P., Joseph, L.N., Szabo, J. & Martin, T.G. (2013). Realising the full potential of citizen science monitoring programs. *Biological Conservation.* Vol 165: 128-138

Tweddle, J.C., Robinson, L.D., Pocock, M.J.O. & Roy, H.H. (2012). 'Guide to citizen science: developing, implementing and evaluating citizen science to study biodiversity and the environment in the UK'. Wallingford: NERC/Centre for Ecology & Hydrology, 29pp.

Van Damme, D. (2013). *Potamopyrgus antipodarum*. The IUCN Red List of Threatened Species 2013. Downloaded on 26th August 2018 from http://www.iucnredlist.org/details/155980/0

Wallace, J.B. & Webster, J.R. (1996). The role of macroinvertebrates in stream ecosystem function. *Annual Review of Entomology.* Vol 41: 115-139

Watts, G., Batterbee, R.W., Bloomfield, J.P., Crossman, J., Daccache, A., Durance, I., Elliott, J.A., Garner, G., Hannaford, J., Hannah, D.M., Hess, T., Jackson, C.R., Kay, A.L., Kernan, M., Knox, J., Mackay, J., Monteith, D.T., Ormerod, S.J., Rance, J., Stuart, M.E., Wade, A.J., Wade, S.D., Weatherhead, P.G. & Wilby, R.L. (2015). Climate change and water in the UK – past changes and future prospects. *Progress in Physical Geography.* Vol 39(1): 6-28

Webster, J.R. (2007). Spiraling down the river continuum: stream ecology and the U-shaped curve. *Freshwater Science.* Vol 26(3): 375-389

West Virginia department of environmental protection. (n.d.) WV Save Our Streams Field Guide to Aquatic Invertebrates Downloaded on 26th August 2018 from: http://dep.wv.gov/WWE/getinvolved/sos/Documents/Benthic/WVSOSAdvanced_MacroGuide.pdf

Young, R.G., Matthaei, C.D. & Townsend, C.R. (2008). Organic matter breakdown and ecosystem metabolism: functional indicators for assessing river ecosystem health. *Journal of the North American Benthological Society.* Vol 27(3): 605-625

Zhang, L. (2017). 'Investigation on the relationship between organic matter decomposition and macroinvertebrate assemblages using colonisation traps'. University College London. Departmant of Geography. (Masters thesis)