Abstract

Theoretical studies of the resilience of ecological systems to environmental change predict that the size distributions of species in ecosystems should have discontinuities that reflect similar discontinuities in ecosystem processes. Body size distributions should have many peaks and troughs (modes) for natural, undisturbed ecosystems, but that as disturbances increases, so the number of modes declines. If so, this prediction has implications for assessing the quality of real ecological systems and has potential for environmental monitoring.

This research was carried out to explore the relationship between water quality and body size patterns in benthic communities in order to establish the potential of size based indicators for assessing environmental condition, as well as testing Holling's (1992) proposition that lumpiness occurs in body size distributions across a broad range of spatial and temporal scales. The invertebrate samples were collected from both lentic and lotic habitats including rivers, ponds and canals in Yorkshire, UK, known to experience different degree of pollution. Five stations on the River Ure system and nine sites on the River Aire were sampled for moving water while eight stations were sampled for static (canals and ponds) habitats in the area South East of York. Community Abundance, BMWP score, water conductivity and ASPT analysis confirmed varying water quality among these sites.

Visual observation of body mass pattern showed skewed distributions towards smaller size classes and most had two very obvious modes at medium and large size classes except for the most polluted habitats. Similarly, statistically rigorous estimates using Kernel Density Analysis (KDE) revealed highly significant relationships between the number of modes and water quality for running water. However, this study was unable to firmly relate discontinuities in body mass distributions to water quality in static water bodies and in fact the relationship appear to be the opposite of those found for moving water. Analysis of the number of gaps, using Holling's (1992) Body Mass Difference Index (BMDI), revealed wide variation in clean and intermediate water quality sites, though the most polluted site had the fewest gaps. However, other disturbed sites had

more gaps and some clean sites had fewer gaps. It is clear that size distributions in benthic communities are lumpy, in the sense that most sites showed more than one mode or many gaps, but the number of gaps (discontinuities) is not always correlated with disturbance, at least for the water quality of freshwater systems.

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Acknowledgements

First of all, I would like to express my sincere gratitude to my supervisor, Professor Dave Raffaelli for his patience, encouragement and support throughout my PhD and my TAC members Dr Piran White and Bryce Beukers-Stewart for their support and advice. I would also like to thank the Environmental Agency Leeds for providing a GQA classification for the River Aire. I am also grateful to Professor Dave Raffaelli, Dr. Mudassar Iqbal and Afia Zia for helping me in the field work and Rebecca Sutton for the technical support in the Lab. I would also like to thank Dr Tsuyuko Yamanaka for her help with my data analysis.

I would like to express many thanks to Higher Education Commission of Pakistan for financial support for my PhD.

Finally, I would like deeply thank to my parents-in-law for their support. Thanks to my family, especially my parents for their encouragement and spiritual support throughout my life. I would like to express my heartiest gratitude to Dr. Mudassar Iqbal (husband) who has been genuine partner throughout my research and Eshal Fatima (daughter) for her love and cooperation.

Declaration

I declare that the work presented in this thesis is my own, where other sources of information have been used, they have been acknowledged. Chapter 4 has been submitted for publication. I completed all of the data collection and analysis myself and the work contained in this thesis has not been submitted for any other degree or award.

Hamida Bibi

Chapter 1

Introduction

1.1 General Introduction

Every living organism depends on water. Two thirds of the Earth is covered with seawater but freshwater habitats are much more restricted and their exploitable volume is only 0.003% of the total, a relatively scarce resource in relation to population growth (Mason, 2002b). Humans continue to pollute water at an alarming rate, which affects humans as well the aquatic biota. As a consequence, more than 20% of the world's population do not have access to safe drinking water and 2.4 billion lack even adequate sanitation. Due to the unhygienic use of water, 25,000 children die every day worldwide and a recent study estimates 2.5 million deaths occur per year from diarrhoea (Kosek, 2003). It is estimated that 88 per cent of diseases are due to the use of unhealthy water, resulting in substantial health and economic losses (WHO, 2004).

Pollutants may be discharged from point sources, for example from factories or sewage treatments works. Alternatively diffuse sources include fertilizer and pesticide applied to crops, and acid precipitation entering water bodies by surface runoff and land drainage. Most pollution is chronic; the effect on the aquatic ecosystem is harmful but not catastrophic and can be reduced. A greater problem now is episodic acute pollution which is unpredictable in space and time and can destroy years of patient and struggling work in reducing the effect of chronic pollution.

Water pollution is problematic due to industrialization and the increasing growth rate of the human population. Wastes from industries and expanding towns have been poured untreated into rivers, causing bad odour as well as epidemic diseases like cholera. Due to these problems, in 1876 the United Kingdom brought in legislation to control water pollution and this was extended to river and coastal water by 1951 (Hammerton, 1995). Currently in the United Kingdom contamination is controlled by specific procedures such as Pollution Prevention and Control (PPC) and environmental regulation like, Best Available Technique (BAT) (Sorrell, 2001). Although water pollution is hazardous to humans and aquatic life, and nutrient pollution such as Phosphorus and Nitrogen comes from agricultural activities, causing harmful algal blooms. Zooplanktons eat toxic algae and these toxins pass up food chains causing illness and sometimes death to higher trophic levels. Algal blooms also cause mortality of fish by encouraging the marine pathogen *Pfiesteria*: in 1991, one million menhaden fish died in North Carolina's Neuse River due to a *Pfiesteria* outbreak (Burkholder *et al.*, 1992). Chemical pollution is also introduced to water bodies from point sources as well as non-point sources like industrial wastes, leaching from contaminated soil and surface runoff, etc. These hazardous chemicals have adverse effects on aquatic life and can alter the ecosystem balance. Thus, the animals and plants which survive in water bodies are a good indicator of water quality.

1.2 The assessment of water quality

Several ways are available to measure water quality. One approach is to take water samples and measure the concentration of different chemicals, either above or equal to a standard, known as a chemical indicator of water quality. Another approach is to examine aquatic organisms, like fish and invertebrates. The presence or absence of healthy populations of organisms within a specific habitat is a sign of particular environmental conditions. These are biological indicators of water quality. Invertebrates have a high capability for accumulating metals in their tissues from their environment, meaning that these contaminants have a large biological half-life, because metals form complexes with organic substances and have tendency to be fixed in the tissue, preventing them from being excreted (Vernberg and Vernberg, 1974). In this way organisms can reflect environmental pollution levels. Whilst chemical analyses give a value at the time the sample is taken, bio-indicators provide information over a longer term basis, because these animals are in that environment for a long time.

Organisms are usually monitored to assess public exposure to materials such as food and potable water. Fish are very important in freshwater monitoring programmes as they are a direct part of human food, but several conditions should be kept in mind before considering organisms as satisfactory sentinels of pollution. Indicator animals should be sedentary so they reflect the pollution level, they must be abundant and easily identifiable, and they should also be large enough to provide concentration levels of pollutants.

The changes induced in the flora and fauna of rivers and streams can be used for biological effects of monitoring of water bodies. Major changes by pollutants occur in macro-invertebrate populations such as mayflies, stoneflies, crustaceans and molluscs etc. These changes in the relative abundance of different groups of macro-invertebrates occur due to the different sensitivity of these organisms to pollutants where pollution sensitive species are replaced by tolerant species and community diversity declines.

Biological surveys are generally undertaken in summer and autumn because of the flows which makes sampling easier and during this period the worst conditions are expected because of the lack of dilution of waste. Samples are taken from rivers, streams or canals and water quality of the site is determined on the basis of relative proportion of various organisms in the sample. These data can also be compared with data taken from clean sites. Using biological approaches, water condition can be categorized into six classes, which includes very good, good, fairly good, fair, poor and bad quality (Logan, 2001).

Physico-chemical evaluation of water quality, in contrast, can be done throughout the year, but the assessment needs constant records of concentration and flow to make a standard basis for quality monitoring. In practice, such chemical testing is impossible for economic, technical and logistic reasons, being highly costly for general water quality assessment when a large number of rivers and streams are to be monitored. A greater number of chemical samples is usually required to provide the same degree of information as just two biological samples per year. For effective chemical analysis, information about the nature of likely pollutants is necessary, making monitoring very difficult. The irregular discharge of pollutants also makes chemical assessment ineffective because sampling may miss important pollution events.

1.3 Biological Monitoring Approaches

To describe the impact of water quality on the biology and ecology of systems a range of techniques can be used such as community diversity indices and biotic indices. A number of biotic indices have been introduced for water quality assessment on the basis of the sensitivity or tolerance of species or taxa to pollution. Indicator species are given a value based on their perceived susceptibility, the sum of which provides an index of pollution for the site. These indices may be qualitative based on the presence or absence of species. Some of these indices are described below.

1.3.1 Saprobic index

This method is qualitative (Pantle and Buck, 1955) and the basis is the presence or absence of indicator species mainly microorganisms such as bacteria, viruses and fungi. The approach describes four stages, oligosaprobic, β mesosaprobic, α mesosaprobic and polysaprobic. The method uses a formula:

$$\textbf{Saprobic index (SI)} = \sum (sh) / \sum h$$

Where (h) express the abundance of different species and s value for Saprobien groups in Table 1.1 below.

Table 1.1 Saprobian Idex of Pantle and Buck (1955).

State	s value		h value
Oligosaprobic	1	Occurring incidentally	1
β – mesosaprobic	2	Occurring frequently	3
α- mesosaprobic	3	Occurring abundantly	5
Polysaprobic	4		

This technique was one of the first developed to detect organic pollutants (Pantle and Buck, 1955) and the ranges of the Saprobic index are given below.

Index range	**Stage**	**Quality**
1.0-1.5	Oligosaprobic	no pollution
1.5-2.5	β – mesosaprobic	weak organic pollution
2.5- 3.5	α- mesosaprobic	strong organic pollution
3.5-4.0	polysaprobic	very strong organic pollution

The technique is not commonly used because of its selective pollutant response and the use microorganisms as indicators.

1.3.2 Index of Biotic Integrity (IBI)

This method combines several metrics of biological indictors in relation to environmental stress into a summary index number and is defined, as "the ability of an aquatic ecosystem to sustain the balanced community organisms, having a species composition, diversity and functional organization comparable to the natural habitat of region" (Barbour and Yoder, 2000). The metrics used for invertebrates in disturbed environmental conditions to determine the IBI are given in Table 1.2.

1.3.3 Diversity Indices

Diversity indices are often used to assess environmental stress. Clean ecosystems are characterized by a large number of indicator species with a balanced community distribution, and with no dominance by a single species. When large numbers of species are present in a community in similar abundance maximum diversity will be obtained. In a disturbed environment species sensitive to a particular stress will be eliminated and as a result tolerant species become dominant in the community. To assess environmental condition using diversity indices, the number of species in a sample and their relative abundance are determined.

Table 1.2. Macro-invertebrates metrics used in calculating the Invertebrate Community Index (ICI).

Category	**Metric**	**Response**
Species composition and richness	➢ Number of species	Decrease
Composition measures	➢ Number of EPT (Ephemeroptera, Plecoptera and Trichoptera)	Decrease
	➢ % EPT	Decrease
Intolerance measure	➢ Number of intolerant taxa	Decrease
	➢ % tolerant taxa	Increase
	➢ % dominant taxon	Increase
Ecological factors		
Feeding measures	➢ % filters	Variable
Habitat measure	➢ % grazer + scrapers	Decrease
Anomalies	➢ Number of clingers in taxa	Decrease
	➢ Proportion with diseases	Increase

The Shannon-Wiener index is a commonly used diversity index, which assumes that individuals are randomly sampled from a large population of species. In this technique, the number of individuals in a given taxon is divided by the total number of organisms in the sample the ratio is then multiplied by the logarithm of that ratio. The formula for the Shannon-Wiener index is given as:

$$H = - \sum Ni/ N \; In \; Ni / N$$

$$\Sigma(Ni/N) \text{ x } \ln(Ni/N)$$

Where N is the total number of species in collected samples and Ni is the number of individuals belonging to *i*th species. Smaller values of H are associated with polluted water whilst higher values are related to cleaner ecosystems.

Seasonally abundant species can result in mis-interpretation of water quality using diversity indices. Sampling method, sampling season, the area sampled and identification level all influence diversity indices (Hughes, 1978). Seasonal variation should be taken into account when comparing data from sites because the seasonal variation recorded may be greater than the differences between sites (Pinder and Farr, 1987). Extreme care is needed in the interpretation of diversity indices and the number of species may give more reliable results (Winner *et al.*, 1975)

Table 1.3. Biological Monitoring Working Party (BMWP) family scores.

Groups	**Families**	**Score**
Mayflies	Siphlonuridae, Heptageniidae, Leptophlebiidae, Ephemerellidae, Potamanthidae, Ephemeridae	10
Stoneflies	Taeniopterygidae, Leuctridae, Capniidae, Perlodidae, Perlidae, Chloroperlidae	
River bug	Aphelocheiridae	
Caddis flies	Phryganeidae, Molannidae, Beraeidae, Odontoceridae, Leptoceridae, Goeridae, Lepidostomatidae, Brachycentridae, Sericostomatidae	
Crayfish	Astacidae	8
Dragonflies	Lestidae, Agriidae, Gomphidae, Cordulegasteridae, Aeshnidae, Corduliidae, Libellulidae	
Caddis flies	Psychomyidae, Philopotamiidae	
Mayflies	Caenidae	7
Stoneflies	Nemouridae	
Caddis flies	Rhyacophilidae, Polycentropidae, Limnephilidae	
Snailsg	Neritidae, Viviparidae, Ancylidae	6
Caddis flies	Hydroptilidae	
Mussels	Unionidae	
Shrimps	Coriphiidae, Gammaridae	
Dragonflies	Platycnemidae, Coenagriidae	
Water bugs	Mesoveliidae, Hydrometridae, Gerridae, Nepidae, Naucoridae, Notonectidae, Pleidae, Corixidae	5
Water beetles	Haliplidae, Hygrobiidae, Dytiscidae, Gyrinidae, Hydrophilidae, Clambidae, Helodidae, Dryopidae Elminthidae, Crysomelidae, Curculionidae	
Caddis flies	Hydropsychidae	
Craneflies	Tipulidae	
Blackflies	Simulidae	
Flatworms	Planariidae, Dendrocoelidae	
Mayflies	Baetidae	4
Alderflies	Sialidae	
Leeches	Piscisolidae	
Snails	Valvatidae, Hydrobiidae, Lymnaeidae, Physidae, Planorbidae	3
Cockles	Sphaeriidae	
Leeches	Glossiphoniidae, Hirudidae, Erpobdellidae	
Hoglouse	Asellidae	
Midges	Chironomidae	2
Worms	Oligocheata (whole class)	1

1.3.4 Biological Monitoring Working Party (BMWP) Index

This technique was described by Hawkes (1998) for general quality assessment of water. In this method scores are given to indicator species on the basis of their perceived susceptibility. A sum of BMWP scores greater than 100 is considered clean while values less than 10 indicated polluted conditions. This technique has weaknesses such as the effect of sampling effort because the larger the sample, the more individuals and species and the method is also influenced by season. To overcome these drawbacks it has became a common procedure to divide the BMWP score by the number of taxa to produce the Average Score Per Taxon. The main difference between these two techniques is that ASPT does not depend on family richness(Armitage *et al.*, 1983). Different families and their BMWP scores are given below in Table 1.3.

1.3.5 Average Score Per Taxon

This technique is independent of seasonal effects and sample size and permits effective comparison between different sites. The approach is quantitative and known as ASPT (Average Score Per Taxon) analysis. The method is considered the most rigorous for the analysis of changes in water quality. In ASPT, identification to family level is enough. Each family present in a sample is given a score between 1 and 10 on the basis of their expected susceptibility to pollution. Species most susceptible to pollution are given the highest score, while tolerant species are given the least score. The total score is then divided by the number of taxa that have been taken into consideration to make the index independent of sample size.

The most useful feature of ASPT is that this technique is not sensitive to seasonal variation and sample size, so any size of samples and at any season can be taken for analysis. ASPT produces consistent results and is an easy technique to assess environmental condition, so that it is routinely used by environmental agencies across the UK. The method explains 65 per cent of the variation, while methods used in other techniques justify results only 22 percent(Armitage *et al.*, 1983). This method has been found to be the best indicator over a range of pollutants such as sewage, insecticides, heavy metals and surface runoff (Sriyaraj and Shutes, 2001)

1.4 Body Size and its Relevance to Disturbance

Body size was first used to explore the dynamics of aquatic ecosystems by Sheldon *et al* (1967a) and the size spectrum may reflect well environmental conditions. Body size is a determinant of a wide range of ecological and evolutionary parameters and the relationship between body size and abundance is a useful way for describing a wide variety of taxa and habitats (Blackburn and Gaston, 1999). The negative relationship between size and abundance at large spatial scales has been widely studied in ecology. Survival of individuals has also been correlated with body size in some studies, species with smaller body sizes experience high survivorship compare to species with larger body sizes and there is negative correlation between population size and probability of extinction (Stanley, 1986) whilst body size and abundance are characteristics that make species sensitive to extinction (McKinney, 1997; Gaston and McArdle, 1994). Shifts in body sizes of animals have been shown to reflect environmental changes like habitat architecture as small animals depend on resources and ecosystem patterns at small scales and large species over large scales (Holling, 1992). Animals develop specific physical and behavioural characteristics to exploit varying environmental texture across scales and show discontinuities in their body size distributions. Body size is an important parameter which determines many aspects of life history like metabolic efficiency, generation time and metabolism (Morse *et al.*, 1988) and there is a positive correlation between metabolism and body size of individuals (Peters, 1983b), whilst body size is also related to reproduction and dispersal (West *et al.*, 1997). Size is also important to consider in avoiding specific sampling biases in ecological and paleontological studies, including sieve size effects (Kidwell, 2002). Differences in body mass patterns have been observed in lake and marine sites, due to two dominant groups (Chironomid midge and Oligochaete worms) which were found most of the lakes (Strayer, 1986). Similarly, the size spectra of benthic communities vary among lakes as a function of water chemistry and the larger fauna (e.g. Cray fish and large Mollusks) are only found in shallow and hard water than deep and soft water (Singer, 1982; Haines, 1981). Methods which use body size distributions to assess particular environmental conditions require fewer data, may be less time consuming relatively inexpensive and repeatable. Due to these suitable characteristics one could prefer to establish size based indicator of environmental assessment.

In the next section I describe various hypotheses to explain variations in body size distributions, the most relevant for this study being Holling's Textural Discontinuity Hypothesis (Holling, 1992).

1.5 Warwick's ABC Index

Other researchers have explored the potential of size-based metrics for pollution monitoring, recognising that long-lived, pollution intolerant species tend to have a large body size, whilst opportunistic species that are either pollution tolerant or which are good at exploiting disturbed systems tend to be of small body size. For instance, Warwick and Clark (1994) developed the ABC (Abundance/Biomass Comparison) index and noted that in marine benthic systems there was often a shift from large bodied species to small bodied species along gradients of organic pollution and that this could be captured graphically by plotting cumulative abundance and cumulative biomass against pollution. In undisturbed communities, the biomass line lies above the abundance curve, due to presence of large bodied organisms, while in moderately disturbed communities, these curves cross over at one or more place. Very disturbed communities are dominated by small bodied individuals and the abundance curve lies entirely above the biomass curve.

Whilst this scheme has a clear logical underpinning, the ABC method sometimes gives a false impression of disturbance due to presence of large number of small individuals in undisturbed sites (Ibanez and Dauvin, 1988), so that it not a reliable method for detecting pollution. Also, Warwick and Clarke (1993) noted that this method is no more sensitive than diversity indices or multivariate analysis at detecting disturbance. Furthermore, in their 1993 paper, Warwick and Clark found that the size effects noted were largely due to replacement of whole phyla along a gradient of increasing pollution, echinoderms disappearing first, then crustaceans and bivalves. In other words, the so-called size effect was more of a phylogenetic one. Given these issues, the ABC approach was not explored here

1.6 Review of Holling's Textural Discontinuity Hypothesis

C.S Holling (1992) wrote an important paper which presented a life time's observations and analysis of complex ecosystems. Central to his thesis is the observation that body size distributions reflect important discontinuities in ecosystem processes and that the number of such discontinuities may be a measure of ecological stability (resilience). My research, in part, explores the evidence for this hypothesis in freshwater systems (moving and still) with different water qualities. Holling's paper is not easy to understand on first reading and this review attempts to set the background for my own work for the reader new to the topic.

Ecosystems are shaped by a few biotic and abiotic processes which exert function at specific spatial and time scales. These small numbers of processes dominate in ecosystems and suppress other processes. They are distributed discontinuously and they create discontinuity across a system by entraining other processes setting the cycle of ecosystem dynamics. Animals living in ecosystems demonstrate this uneven distribution of processes by showing gaps in their own processes which are reflected in their body size distributions. This unevenness (presence of gaps) provides evidence of "a lumpy architecture" of specific landscapes where these animals live, the body mass clumps being restricted and controlled by the structure and texture of the landscape. This hypothesis is therefore known as the Textural Discontinuity Hypothesis (Holling, 1992).

There are three processes (micro-scale, macro-scale and meso-scale) that generally control habitat architecture, creating different ranges of scale in time and space. Vegetative processes or micro-scale processes act at the micro level, centimeter to meters in space and at days to decades in time, determining plant growth and soil structure. Geomorphologic processes are macro-scale and work at kilometers scale in space and centuries to millennia in time. Disturbance processes like fire, insect outbreaks and plant diseases dominate over meters to kilometers with time scales of years to decades and are meso-scale. Human activities also act as meso-scale processes.

Studies on mammals in short grass prairie and birds of boreal region forests by Holling (1992) show a clumpy structure in their body masses distributions. Also, the number of clumps and gaps are the same and occur at the same body masses in the two different habitats indicating that processes which create habitat structure are common for both grassland and forest. (Birds and mammals utilize resources from habitats as a function of their size).

To check that these features of size distributions are as a result of ecosystem organization, rather than artifacts, critical studies of the model were carried out. In Holling's model variables behave differently, with a number of cycles such as a fast variable 3-5 year cycle caused by interactions among leaves, parasites and insect defoliators (McNamee, 1979); an intermediate periodicity of 10-15 represented by interactions between defoliation and time of recovery of foliage quantity and quality with slow growth and less mortality of plants; a cycle with a periodicity 35-40 years is the interaction between fast and slow variables and agents like birds; the ≥ 80 year cycle is set by long-lived trees caused by wind, diseases and insects. Some authors suggest that these cycles occurring in different periods are present within an ecosystem (Royama, 1984) having both spatial and temporal aspects, which strongly indicates that an ecosystem is shaped by a small number of such processes which dominate and entrain one another, creating discontinuities which define specific periodic, spatial patterns and frequencies. Study on keystone species (Paine, 1966), also showed size patches and structural distributions of species within a community. There are several explanations for these features, which Holling (1992) lists as the following basic propositions.

1.6.1 The Extended Keystone Hypothesis

According to this hypothesis, all systems are controlled by a small number of key plants, animals and abiotic functions operating over spatial scales of centimeters to kilometers and from months to centuries of temporal scale. If so, each ecosystem should have these dominant processes with discontinuous frequencies, and these processes can occur as a result of evolutionary adaptation or because of cycling of variables in the structuring of the ecosystem.

1.6.2 The Entrainment Hypothesis

In this hypothesis biotic processes occur at regular intervals within ecosystems by forming clusters, reflecting generation times of the species involved. In this regard forest insects present a small set within a hierarchy associated with needles and parasites as faster variables, crown and insectivorous birds are slower variables and tree species are considered as the slowest variables and coarser patches.

In some studies (MacArthur and MacArthur, 1961), the biotic structure of an ecosystem is shaped by processes such as species distributions associated with vegetation or habitat structure. Other observations indicate that plant species determine community composition rather than the physical architecture of ecosystem (Wiens and Rotenberry, 1981).

All the above can generate the lumpiness seen in body mass distributions but there are four major competing hypotheses that Holling (1992) considers.

1.6.3 Textural Discontinuity Hypothesis

TDH assumes that body mass distributions of species are bioassays of landscape structure because they show correlation with texture and the hierarchic nature of the landscape/seascape. According to this hypothesis animals should express the discontinuous structure of the landscape where they live by showing discontinuity in their body mass distributions, reflecting their scale of foraging and searching and other behavioral choices.

1.6.4 Limited Morph Hypothesis

This hypothesis proposes that species are constrained to a limited range of size and functions because they have limited life manner, such as locomotry modes. The limited ability of species to disperse due to geographical boundaries may result because limited groups of species are present in a community. These function over a limited range of size, but there are a few species that form body size clumps constrained by life forms. Some studies suggest that gaps in body mass distributions are due to the different capacity of species to locomote across a landscape and this can be seen where more species are larger than smaller (Wiens and Rotenberry, 1981).

1.6.5 Historical Hypothesis

Species body sizes are constrained by evolution and adaptation to form a complex system. Processes such as different evolutionary histories constrain the distribution of body mass within taxa which may result in multimodality (Smith *et al.*, 2004). The area's ancestral characteristics limit the body mass evolution of offspring within taxa. In most of the world's mammals body size distributions are right skewed, and this is phylogenetically independent (Gardezi and Silva, 1999). In other studies it has been found that there is no relation between body masses and taxonomy (Sendzimir *et al.*, 2003).

1.6.6 The Trophic Trough Hypothesis

The body size of species controlled by trophic interfaces creates clumpy structures in their distribution patterns. Resource limitation causes changes in body mass distributions and secondary modes produce a distribution towards large body size contrary to the right skewed distribution described above. Predators select prey based upon size and the species with smaller body sizes are selectively removed from the community which leaves gaps in body mass distributions and multimodality appears. The lack of multimodality among species that do not feed on the same resource suggests that competition between species for similar resources may cause discontinuities in these size distributions (Stubblefield *et al.*, 1993).

1.7 Textural discontinuity and ecological stability

The Textural Discontinuouity Hypothesis provides a good connection between community structure and ecosystem dynamics by suggesting that processes which cause discontinuous changes over time and space reflect the hierarchical physical structure of system. There are three scales at which dominant processes operate:

The microscale, determined by vegetation, the mesoscale, determined by environmental processes, and the macroscale, controlled by evolutionary processes. At the microscale, the dominant processes make structures similar in all ecosystems because these processes have the same geometrical structure therefore small body mass clumps are similar from one landscape to another. In contrast, mesoscale and macroscale processes are landscape-specific and body mass clump structures differ accordingly. Mesoscale processes are a spreading feature and assist rapid changes in the ecosystem. Within landscapes these scales group together to form cycles such as birth, growth, energy storage and regeneration at that scale, like the yearly needle cycle, crown of foliage over a decade and tree gaps over centuries.

If such discontinuities are a feature of natural systems, then we would expect body size distributions to be similarly clumped with gaps. Gaps may therefore be a sign of a "healthy" ecosystem. One of the aims of this research work is to see how the number of gaps varies in habitats which can be ranked according to their "healthiness"

Holling claimed that the degree of heterogeneity created by patchiness in space and time creates a dynamic mosaic across a range of scales which is related to the degree of stability of an ecosystem based on the following arguments. Because this patchiness is such a common feature of natural systems, it seems likely that it is related to the long-term persistence of ecosystems: if patchiness was a destabilising feature, then how could it have persisted? Related to this argument are the long-term observations of Holling on the dynamics of spruce communities within the great Boreal Forest of North America. These observations lead to the development of Resilience Theory and related concepts, such as adaptive cycles of ecosystem processes operating at specific scales of space and time (Figure 1.6.1) and which consist of 4 phases: exploitation, conservation, release and re-organisation. Raffaelli and Frid (2010) and (Walker and Salt, 2006) discuss the importance of these cycles for long term stability of entire ecosystems, but in the present context of this thesis, it should be noted that adaptive cycles operate at all spatial and temporal scales in a forest, from individual leaves growing and dying on trees, to the life and death of entire stands of trees and forests (Figure 1.2). It is important for forest management that cycles at different scales do not become synchronised over large areas, because this would cause the collapse of the forest, as all cycles would enter the collapse or release phase (Figure 1.2) at the same time. In other words, the number of separate cycles operating at different time and space scales becomes fewer, perhaps merging into one big cycle so the whole system collapses. Thus, Holling argued that a healthy ecosystem which persists for long time periods should be characterised by many different cycles operating at different scales and which are out of phase. Unhealthy systems have fewer cycles and are less likely to be resilient to external change. In other words, those ecosystems which are disturbed or less stable and resilient should have a lower degree of heterogeneity in their processes across a range of scales. (Interestingly, it is frequently observed for mathematical models of interacting systems (networks) that they self-organise over time with some network nodes becoming linked to many other nodes, whilst other nodes remain weakly connected, generating a heterogeneous or “lumpy” network).

Direct tests of these ideas are difficult but can be tested indirectly using surrogates of ecosystem processes, the body sizes of the organisms in the ecosystem; because body sizes are a reflection of processes operating at different scales (see discussion above). Thus, in Holling’s plots of the adult body sizes of birds and mammal species from North American grasslands and forests, many modes are apparent which he claimed were associated with ecosystem processes operating at specific scales (although the identities of these processes were not known, only suspected). The regions between modes, the so-called “gaps”, were claimed to represent the discontinuities between ecosystem processes. Holling further argues that these gap regions would be the most susceptible to disturbance and where species losses would be most likely. Other members of his research group, notably Craig Allen, took these ideas further by examining how the body sizes of invasive species in the Florida Everglades (Allen *et al.*, 1999) and elsewhere fitted in to the existing body size distributions of the community being invaded, reviewed in (Allen *et al.*, 2006) and references therein. They found that invasive species tended to have body sizes that were immediately adjacent to the gap regions, and that species which were lost due to disturbance were close to these gaps, consistent with Holling’s predictions. (Raffaelli *et al.*, 2000) also tested this idea for a marine intertidal community and found that the body size distributions were multimodal (as previously suggested by (Schwinghamer, 1981b) for marine sediments and that at least one kind of disturbance, which they applied experimentally, organic enrichment, had the greatest impact on body sizes in and adjacent to one the troughs between modes.

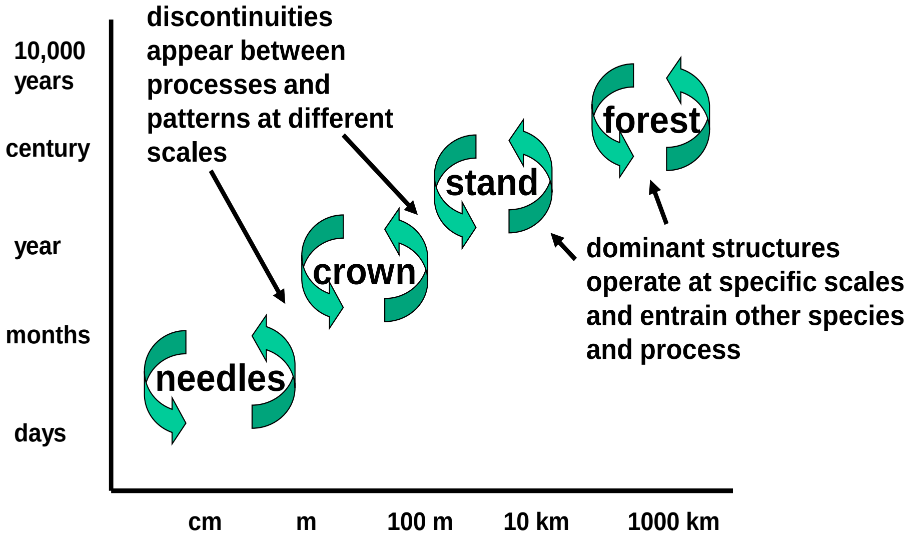


Figure 1.1: Patchiness occurs at a range of spatial and temporal scales in nature, as shown in this example of a spruce forest ecosystem. In such systems, dominant structures (from needles to forests) operate over different spatio-temporal scales. The cycles of life and death for each of these structures may follow adaptive cycle dynamics (see Fig 1.2), and these may entrain other ecological processes. From Raffaelli and Frid (2010)

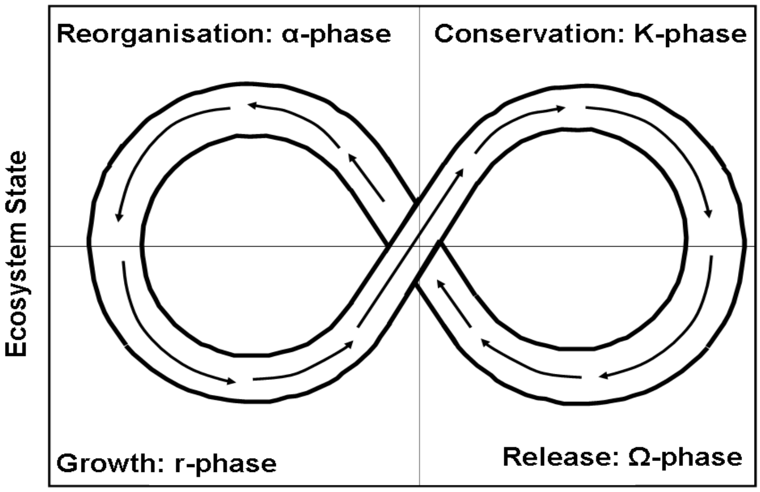


Figure1.2: The adaptive cycle view of ecosystem development and change. In this perspective, collapse of the system is inevitable, whereupon the system components may re-assort and begin development again as a broadly similar system or one which is very different. From Raffaelli and Frid (2010).

From the above, it can be seen that one of the effects of disturbance on body size distributions may be first to deepen the troughs between modes (make them more pronounced), if that disturbance only leads to species loss, as in the case of eutrophication (Figure 1.3). However, the response of ecological systems to eutrophication or enrichment is not usually a “simple, or monotonic”. At moderate enrichment, there may be an increase in the abundance of all species (and body sizes), but at higher levels of enrichment, the positive effects may be overtaken by the negative effects as some species intolerant of low oxygen concentrations brought about by a high BOD may be excluded and smaller taxa which are more tolerant of pollution dominate. In such cases, the body size distributions may at first maintain their structure and modality, but as pollution increases, the larger taxa will become excluded and the size structure becomes more skewed towards smaller animals altering modality. Such changes in body size distributions are well-documented in aquatic communities as empirical observations (Warwick, 1984)), but their consequences for, and relationships with, changes in ecosystem processes at different scales have not been explored in the context of Holling’s theories.

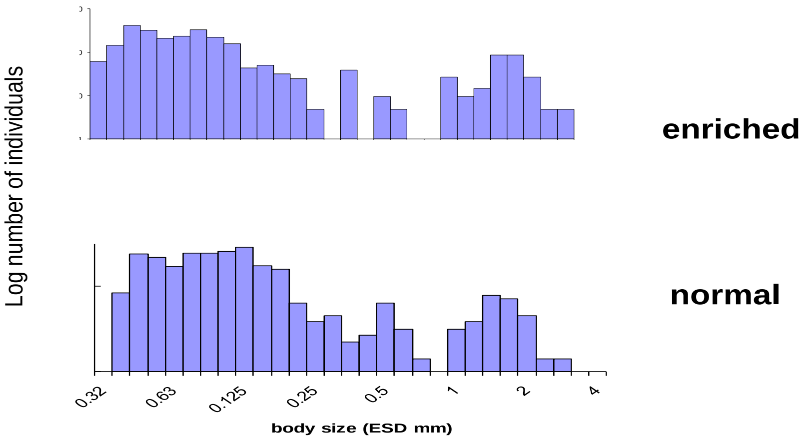


Figure 1.3: The effect of pollution on body mass distributions on benthic communities Raffaelli et al (2000).

As far as I am aware, there have been no previous experimental or empirical published studies looking at the degree of modality in body size distributions (as a reflection of the heterogeneity in ecosystem processes) and disturbance (in this case pollution). Raffaelli, Smart and White (in preparation) compared the body size distributions of the mammal fauna of the UK from the Mesolithic (undisturbed) with the present day (very disturbed) and found a reduction in modality from 3 to 1, due to a combination of the loss of large predatory herbivores and predators (megafauna) and the infilling of the Mesolithic toughs by invasive and introduced species), but that study was restricted to mammals (a single taxon). A similar loss of larger body size modes (reduction in modality) can be seen in comparisons of the Pleistocene fauna of North America with the present day, due to over-exploitation of the megafauna by early man (Smith *et al.*, 2004), but that research was not placed in the context of Holling's hypothesis. This thesis therefore represents the first empirical test of this aspect of Holling's theory.

1.7.1 Techniques used to find out gaps and clumps

To test these hypotheses, precise data are required which should not give any false appearance of pattern. Species were chosen in Holling's original research with adult body sizes from a range of taxonomic groups which are considered ideal to test the hypothesis controlling for confounding effects of morphology and evolutionary history. The first thing to keep in mind here is the clumpy representation of data. Inspection by eye of data body size distribution of adult birds of boreal forests and of mammals of short grass prairie shows cumulative distributions of body masses that are identical in terms of locations of clumps and gaps. However, interpreting by eye is very subjective. To present the data more objectively a technique termed the "Body Mass Difference Index" (BMDI) was applied to focus more on gaps than on clumps. The Body Mass Difference Index is the difference between body masses of neighboring species that are arranged in ascending order of body size. To remove the effect of increasing likelihood of gaps with increasing body mass, a value for γ a coefficient (1.1 for mammals and 1.3 for birds) is applied to the formula which becomes,

$$\mathbf{BMDI = (M_{n+1} - M_{n-1})/M_n)^{\gamma}}$$

The results of applying this model to the Holling's birds and mammals data, reveals discontinuities, that is, a number of clumps and gaps. These were taken as "statistically significant" if there are several consecutive differences in less than 1 SE of the mean differences, then this defines a clump. Similarly, if several consecutive differences are higher than 2SE, this is a gap, according to Holling (1992).

To test if body sizes are statistically more lumpy than expected by chance, a bootstrap statistical technique can be used by representing continuous samples with the same size, mean and standard error of the observed distribution. When applied to the distribution of birds and mammals, the result for birds in forests shows no lumpy structure, but species of prairie grass show strong evidence of discontinuities. To test this further, bird data were converted to a lognormal distribution and one thousand mimic distributions were made and analyzed for gaps and clumps using the BMDI criteria above (Hollings, 1992). The results reveal clumps as in the original real data with a probability of 0.005 compared to those distributions derived by this bootstrapping technique (random distribution). So it appears from these tests that the gaps are real and that the body mass distributions are discontinuous or lumpy.

1.7.2 Kernel Density Estimates (KDE)

Silverman's bump hunting technique was used in the present study to describe the body mass patterns explicitly because gap analysis by BMDI was criticised by Manly (1996), who found multiple peaks in Holling's datasets using Silverman's bump hunting technique. Thus, both methods, BMDI and KDE, were used to explore lumpiness in body mass distributions, but it should be noted that they have different strengths and weaknesses. KDE is a robust statistical procedure which finds modes, the location of which can be derived and hence also the location of the troughs between adjacent modes. BMDI is less rigorous and the significance of gaps (and clumps) is determined by Holling's rule-of-thumb (-1SE or +2SE). Gaps are not the same as troughs necessarily and KDE is known to be more conservative than BMDI. By using both techniques, I was able to explore the potential of each for detecting lumpiness and for examining the relationship between environmental disturbance and ecological resilience (reflected by the number of gaps or troughs).

The Kernel Density Estimate (KDE) approach for detecting modes in distributions was put forward by Rosenlatt (1956), Whittle (1958) and Prazen (1962), but the approach did not find widespread application until adequate computing power became available. Silverman described a test where the null hypothesis is that a distribution has n modes, versus the alternative that it has n+1 or more modes (Silverman, 1981). He combined kernel density estimation and smoothed bootstrap resampling to produce a critical bandwidth for modality. This smoothed bootstrap test for modality is considered conservative even in large samples (Peter and Matthew, 2001; Seaman and Powell, 1996) and produces an objective statistical test for the presence of clumps in body mass distributions. In this bump hunting technique clumps are considered to occur wherever a peak or mode is present in a density function associated with that body mass distribution. The density function is the probability density function for body mass, which is produced by kernel density estimation from the distribution of body masses within the sample data and produces a smoothed version of the frequency histogram. The density estimation analysis is a generalization of a visual histogram and gives a valuable indication of skewness and modality in the data.

KDE was carried out using advanced programming language ‘R’ (R Core Development Team, 2004) developed by Dr Jim Smart of this department used here to describe an implementation of bump hunting technique. The analysis determines the clumps for similar body mass across different taxa. However, the frequency histogram produced from given data depends on the bandwidth used. The higher bandwidths produce more smoothing into the kernel density estimates and using a narrower bandwidth reduces the intervening gaps of more than a certain size between datapoints and can no longer be smoothed completely when individual kernels are summed together. Thus, the established critical bandwidth for successive modalities is the minimum bandwidth which can be used to produce a number of peaks from a dataset. The technique is based on bootstrapping since a non parametric density function developed from the data themselves and is regarded as the best available estimate of the underlying distribution and the modality developed from the data is considered to provide the best (probably the only) estimate. The underlying modality of the body mass density function is analysed to determine if an ‘excessive’ amount of smoothing is required to produce a K-modal density function from the data as opposed to a density function with (K+1) or more.

The statistical assessment for ‘excessive’ k-modal smoothing is produced by examining the properties of smoothed bootstrap replicates drawn from the k modal critically smoothed kernel density estimates produced from the original data. Smoothed bootstrap re-sampling is designed to draw replicates from particular kernel density estimates as well as appropriate selection of smoothing bandwidth (Efron and Tibshirani, 1993). A 5% significance level was applied to the estimate of the density function underlying a body mass distribution.

1.8 Aims

The overall aim of this thesis is to investigate benthic communities in freshwater (both moving and static) in term of their body mass distributions and water quality. The specific aims of the project were to:

1. Construct reliable body size spectra for invertebrate communities from different water bodies in Yorkshire, varying in quality, in both static and flowing systems.
2. Analyse those spectra for the occurrence of modality and gaps, using KDE and BMDI.
3. Examinine the relationship between discontinuities in body mass distributions (number of modes and gaps) and water quality (disturbance).
4. Assess the influence of season, sampling techniques and habitat type (moving and static) on body mass distributions.
5. Explore the various hypotheses above for the explanation of multimodality in these systems.

1.8 References

Allen, C. R., Forys, E. A. and Holling, C. S. (1999). Body mass patterns predict invasions and extinctions in transforming landscapes. *Ecosystems* 2(2): 114-121.

Allen, C. R., Garmestani, A. S., Havlicek, T. D., Marquet, P. A., Peterson, G. D., Restrepo, C., Stow, C. A. and Weeks, B. E. (2006). Patterns in body mass distributions: Sifting among alternative hypotheses. *Ecology Letters* 9(5): 630-643.

Armitage, P. D., Moss, D., Wright, J. F. and Furse, M. T. (1983). The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running-water sites. *Water Research* 17(3): 333-347.

Barbour, M. T. and Yoder, C. O. (2000). The multimetric approach to bioassessment, as used in the United States of America. In assessing the biological quality of fresh waters: Rivpacs and other techniques (wright, j. F., sutcliffe, d. W. & furse, m. T., eds), . *Freshwater Biological Association, Ambleside, Cumbria, UK*: 281–292.

Blackburn, T. M. and Gaston, K. J. (1999).The relationship between animal abundance and body size: A review of the mechanisms. In *Advances in ecological research*, Vol. Volume 28, 181-210 (Eds A. H. Fitter and D. Raffaelli). Academic Press.

Burkholder, J. M., Noga, E. J., Hobbs, C. H. and Glasgow, H. B. (1992). New "Plantom" Dinoflagellate is the causative agent of major estuarine fish kills. *Nature* 358: 407-410.

Efron, B. and Tibshirani, R. J. (1993). *An introduction to the bootstrap.* Chapman and Hall, London, UK.

Gardezi, T. and Silva, J. d. (1999). Diversity in relation to body size in mammals: A comparative study. *The American Naturalist* 153(1): 110-123.

Gaston, K. J. and McArdle, B. H. (1994). The temporal variability of animal abundances: Measures, methods and patterns. *Philosophical Transactions: Biological Sciences* 345(1314): 335-358.

Haines, T. A. (1981). Acidic precipitation and its consequences for aquatic ecosystems: A review. *Transactions of the American Fisheries Society* 110(6): 669-707.

Hammerton, D. (Ed) (1995). *Water pollution. In ralph c. Kirkwood, anita j.Longley. Clean technology and environment.*

Hawkes, H. A. (1998). Origin and development of the biological monitoring working party score system. *Water Research* 32(3): 964-968.

Holling, C. S. (1992). Cross-scale morphology, geometry, and dynamics of ecosystems. *Ecological Monographs* 62(4): 447-502.

Hughes, B. D. (1978). The influence of factors other than pollution on the value of shannon's diversity index for benthic macro-invertebrates in streams. *Water Research* 12(5): 359-364.

Ibanez, F. and Dauvin, J. C. (1988). Long-term changes (1977 to 1987) in a muddy fine sand abra alba - melinna palmata community from the western english channel: Multivariate time series analysis. *Marine Ecology Progress Series* 49: 65-81.

Kidwell, S. M. (2002). Mesh-size effects on the ecological fidelity of death assemblages: A meta-analysis of molluscan live-dead studies. *Geobios* 35(Supplement 1): 107-119.

Kosek, M. B., C. Guerrant, R.L (2003). The global burden of diarrhoeal disease, as estimated from studies published between 1992 and 2000. *Bull World Health Organ* 81: 197-204.

Logan, P. (2001). Ecological quality assessment of rivers and integrated catchment management in england and wales. *J. Limnol.* 60: 25-32.

MacArthur, R. H. and MacArthur, J. W. (1961). On bird species diversity. *Ecology* 42(3): 594-598.

Manly, B. F. J. (1996). Are there clumps in body-size distributions? *Ecology* 77(1): 81-86.

Mason, C. F. (2002). *Biology of fresh water pollution.*: Pearson Educated Limited.

McKinney, M. L. (1997). Extinction vulnerability and selectivity: Combining ecological and paleontological views. *Annual Review of Ecology and Systematics* 28: 495-516.

McNamee, P. J. (1979). A process model for eastern black-headed budworm. *Can. Ent.* 111: 55-66.

Morse, D. R., Stork, N. E. and Lawton, J. H. (1988). Species number, species abundance and body length relationships of arboreal beetles in bornean lowland rain forest trees. *Ecological Entomology* 13(1): 25-37.

Paine, R. T. (1966). Food web complexity and species diversity. *The American Naturalist* 100(910): 65-75.

Pantle, R. and Buck, H. (1955). Die biologische uberwachung der gawasser und die darstellung der ergebnisse. Gas-u- wasserfach. 96.

Peter, H. and Matthew, Y. (2001). On the calibration of silverman's test for multimodality. *Statistica Sinica* 11: 515-536.

Peters, R. H. (1983). The ecological implication of body size. *Cambridge University Press,Cambridge*.

Pinder, L. C. V. and Farr, I. S. (1987). Biological surveillance of water quality. Temporal and spatial variation in the macro-invertebrates fauna of the river frome, a dorset chalk stream. *Arch. Hydrobiology.*, 109: 321-331.

Prazen, E. (1962). On the estimation of a probability density and mode. *Annals of Mathematics and Statistics* 33: 1065-1076.

R Core Development Team, R. (2004).A language and environment for statistical computing. In *R Foundation for Statistical Computing, Vienna, Austria* *http://www.R-Project.org*.

Raffaelli, D., Hall, S., Emes, C. and Manly, B. (2000). Constraints on body size distributions: An experimental approach using a small-scale system. *Oecologia* 122(3): 389-398.

Raffaelli, D. G. and Frid, C. L. (Eds) (2010). *Ecosystem ecology: A new synthesis* Cambridge University Press

Rosenblatt, M. (1956). Remarks on some nonparametric estimates of a density function. *The Annals of Mathematical Statistics* 27(3): 832-837.

Royama, T. (1984). Population dynamics of the spruce budworm choristoneura fumiferana. *Ecological Monographs* 54(4): 429-462.

Schwinghamer, P. (1981). Characteristics size distribution of integral benthic communities. *Candian Journal of Fishries and Aquatic Science* 38: 1255-1263.

Seaman, D. E. and Powell, R. A. (1996). An evaluation of the accuracy of kernel density estimators for home range analysis. *Ecology* 77(7): 2075-2085.

Sendzimir, J., Allen, C. R., Gunderson, L. and Stow, C. A. (Eds) (2003). *Implications of body mass pattern: Linking ecological structure and processes to wildlife conservation and management. In: Landscape ecology and resource management: Linking theory with practice.*: Island press, Washington, DC, USA.

Sheldon, R. W. and Parsons, T. R. (1967). A continuous size spectrum for particulate matter in the sea. *J. Fish. Res. Board Canada.* 24: 909-915.

Silverman, B. W. (1981). Using kernel density estimates to investigate multimodality. *Journal of the Royal Statistical Society. Series B (Methodological)* 43(1): 97-99.

Singer, R. (1982). *Effects of acidic precipitation on benthos. Pages 329-363 in f.M. D'ltri(editor). Acidic precipitation: Effects on ecological system.* . Ann Arbor Scientific Publisher, Ann Arbor, Michigan.

Smith, F. A., Brown, J. H., Haskell, J. P., Lyons, S. K., Alroy, J., Charnov, E. L., Dayan, T., Enquist, B. J., Ernest, S. K. M., Hadly, E. A., Jones, K. E., Kaufman, D. M., Marquet, P. A., Maurer, B. A., Niklas, K. J., Porter, W. P., Tiffney, B. and Willig, M. R. (2004). Similarity of mammalian body size across the taxonomic hierarchy and across space and time. *The American Naturalist* 163(5): 672-691.

Sorrell, S. (2001). The meaning of BATNEE: Interpreting excessive costs in UK industrial pollution regulation. *Journal of Environmental Policy & Planning* 4(1): 23-40.

Sriyaraj, K. and Shutes, R. B. E. (2001). An assessment of the impact of motorway runoff on a pond, wetland and stream. *Environment International* 26(5-6): 433-439.

Stanley, S. M. (1986). Population size, extinction, and speciation; the fission effect in neogene bivalvia. *Paleobiology* 12(1): 89-110.

Strayer, D. (1986). The size structure of a lacustrine zoobenthic community. *Oecologia* 69(4): 513-516.

Stubblefield, J. W., Seger, J., Wenzel, J. W. and Heisler, M. M. (1993). Temporal, spatial, sex-ratio and body-size heterogeneity of prey species taken by the beewolf philanthus sanbornii (hymenoptera: Sphecidae). *Philosophical Transactions: Biological Sciences* 339(1290): 397-423.

Vernberg, F. J. and Vernberg, W. B. (Eds) (1974). *Pollution and physiology of marine organisms* Academic Press, New York.

Walker, B. and Salt, D. (Eds) (2006). *Resilience thinking: Sustaining ecosystems and people in a changing world.*

Warwick, R. M. (1984). Species size distributions in marine benthic communities. *Oecologia* 61(1): 32-41.

Warwick, R. M. and Clarke, K. R. (1993). Comparing the severity of disturbance: A metaanalysis of marine macrobenthic community data. *Marine Ecology Progress Series* 92: 221-231.

Warwick, R. M. and Clarke, K. R. (1994). Relearning the abc: Taxonomic changes and abundance/biomass relationships in disturbed benthic communities. *Marine Biology* 118(4): 739-744.

West, G. B., Brown, J. H. and Enquist, B. J. (1997). A general model for the origin of allometric scaling laws in biology. *Science* 276(5309): 122-126.

Whittle, P. (1958). On the smoothing of probability density functions. *Journal of the Royal Statistical Society. Series B (Methodological)* 20(2): 334-343.

WHO (2004). Burden of disease and cost-effectiveness estimates.

Wiens, J. A. and Rotenberry, J. T. (1981). Habitat associations and community structure of birds in shrubsteppe environments. *Ecological Monographs* 51(1): 21-41.

Winner, R. W., Van Dyke, J. S., Caris, N. and Farrel, M. P. (1975). Response of the macroinvertebrate fauna to a copper gradient in an experimentally-polluted stream. . *Verh. Internat. Verein. Limnol.* 19: 2121–2127.

Chapter: 2

General Methodology

2.1 Introduction

This chapter reviews the methods used in thesis, providing detail about the techniques and procedures applied in field and in the lab for collecting, sorting and analysis of benthic macro-invertebrates.

In section 2.2, I discuss the way samples were collected, the equipment used, the procedure exercised in field and the number of areas chosen for sampling. Different approaches used for sample collections such as Surber and kick sampling, and the net sampling used for lentic habitats, are also reviewed in this section. Section 2.3, explains the method and apparatus used for the separation and treatment of benthic fauna. In section 2.4, I review taxonomic groups recorded. Section 2.5 discusses the method used for body mass determination of the benthic fauna in this research and the equations from literature and other allometric relationship used to find the dry mass of benthic communities.

2.2 Sample Collection

There are a number of methods that can be used to sample invertebrates in fresh water, but in this research programme the two most appropriate methods, Surber and kick sampling, were used. The Surber sampler is a quantitative way of collecting invertebrates from stream beds and is suitable for shallow water (Figure 2.1a). Samples from River Ure and River Aire were collected using the Surber sampler, with an attached quadrat of fixed area of 0.1 m^2, equipped with a 200 μm mesh net. The frame was placed in the water in such a way that the mouth of the net was perpendicular to, and facing into the flow of water

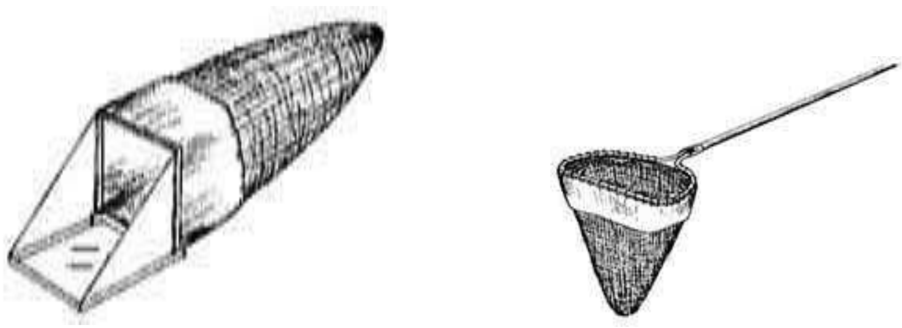


Figure 2.1 Surber sampler (a) and kick sampling net (b)

Source:http://www.google.co.uk/images?hl=en&q=Surber%20sampler&um=1&ie=UTF-8&source=og&sa=N&tab=wi

Within the metal frame (sampled area) stones were carefully pick up and gently rubbed in front of the net to remove any attached animals. Then hands were dug into the substrate to a depth of about 4cm so that the current carried all loosened material into the net. The side wings of the net decrease the chances of animals being lost. When the water ran clear (3-5 minutes), sampling was stopped.

The typical way of collecting samples from streams is by kick sampling, using a standard pond net. It is the simplest way of sampling invertebrates from the bottom of rivers and streams. In this method of sampling a standard pond net with a pore size 500 μm a frame height of 200 mm and frame width of 300 mm, attached with 1.5m handle was used (Fig 2.1b). The net is held vertically in front of the person who faces downstream, with the bottom of the net placed against the substratum. The substratum immediately upstream of the net is then forcefully disturbed with the feet for 3-5 min and any dislodged invertebrates are carried into the net. The method is semi-quantitative but sites can be compared by kicking for a fixed period of time. The method is comparatively rapid and an easy way for relative abundance estimates. For deep locations, like the River Foss, where the Surber technique cannot be used, it is the only way to collect invertebrates in a standardised and unbiased way. Five replicates were collected by Surber and kick sampler, following a zigzag path across the stream bed from each site to make sure that all habitats were sampled. Each replicate was separately transferred to a plastic bag and labelled. The material was brought to the laboratory and transferred to a jar and labelled for each site and preserved in 70 % alcohol.

Kick sampling and Surber sampling work well for flowing streams, but are not appropriate for the static water bodies sampled in chapter 5 (ponds and canals). A standardised approach was devised for the static water bodies. The pond net described above was fitted with a 200µm mesh and placed on the substratum in the shallows (0.5m depth) and dragged for 5 metres to sample an area of 1.5m^2. The volume of material collected in this way was so great that only one sample (replicate) was collected at each of the static sites.

2.3 Separation of Animals

Individuals were separated from debris and sediments by using a 500µm mesh sieve, placed in a white tray following elutriation of the original sample with tap water until any coarse sediment remained. To make animals visible in the tray, the material retained on the sieve was suspended within water in the tray. Large animals were separated first and the material was then very carefully screened for smaller animals which were transferred into vials, labelled and preserved with 70% ethanol. Any remaining sediment was again preserved in case inspection revealed tiny individuals that remained in the substrate.

2.4 The Invertebrates Recorded

Invertebrates were identified to the lowest possible taxonomic level with the help of a range of identification keys (Croft, 1986; Elliott, 1988; Bass, 1998), at least to family level, and counted for each sample. A total of 51 taxonomic groups were recorded across all data sets. The composition of taxa was different between flowing and static communities. For instance, in ponds and canals Corophiida, Argulidae and Vellidae were recorded, but these were not present in stream communities, and several stream taxa were not found in ponds. Some individuals were unidentifiable, mainly because of physical damage, and therefore these animals were excluded from the analysis. The characteristics and images of invertebrates found are summarised below.

2.4.1 Mayflies

These are aquatic insects belong to order Ephemeroptera. The immature stages of these insects are called nymphs. The larvae can be collected from shallow water with a Surber sampler or with the pond net. Mayflies require clean, oxygenated and unpolluted habitats and therefore can be used to assess water quality. They are unique among the insects in having two winged adult stages. As adults they generally live from 1 to 2 hours to a few days and spend most of their life either as eggs or nymphs. Mayfly eggs have a variety of attachment structures that enable them to adhere to submerged objects or to the substrate. Most nymphs hatch at temperatures in range of 3-21˚C and show considerable diversity in habitat and appearance. The different mayfly nymphs recorded are as below.

Baetidae. The larvae are mostly found in sandy bottoms and slow flowing sections of streams (Elliott, 1988). These insects have seven pairs of plate like-gills with one plate larger than the other, two antennae either close together or wide apart, head and thorax laterally compressed or wide bodied in appearance with no compressed head and thorax. Like all mayflies, Baetidae have 3 tails, and the middle tail is slightly shorter than the outer tails. The recorded body mass range for this family is 1-3mg.

Heptagenidae. These larvae are found in stony habitats, the gills are usually flat with or without dots, and have a round or rectangular shape. The shape of the head capsule is round or rectangular. The body mass range is recorded as 0.06-0.8mg.

Caenidae. These larvae are normally found in silty bottoms. The head of these larvae has three prominent ocellar tubercles, the claws are robust and bent at an obtuse angle, and fine hairs cover most of the body. The body mass range is 0.05-2mg.

Ephemerellidae. These larvae are found in fast flowing streams, pigments on gills make it easier to identify this group, and the pigmentation is oval shaped. The body mass range reported is 0.02-2mg for these mayflies..

Potamanthidae. These larvae are found in stoney and sandy bottoms. They are small and have a sub-cylindrical body with oriented gills. The highest densities of Potamanthidae are found during late summer and fall (Munn and King, 1987). The body mass range recorded is 0.06-2mg.

Ephemeridae. These larvae have abdominal marks and are found in muddy substrata (Elliott, 1988). The body mass range recorded is 0.07-3mg. The images of all these families are show below (Fig 2.2).

Baetidae (5mm) Ephemerellidae (4mm) Caenidae (4mm)

Ephemeridae (5mm) Potamanthidae (4mm) Heptageniidae (3mm)

Figure 2.2 The mayfly families recorded in this study, the scale line given in mm.

(Source:http://www.google.co.uk/images?um=1&hl=en&tbs=isch%3A1&sa=1&q=Mayflies+families&aq=f&aqi=&aql=&oq=&gs_rfai)

2.4.2 Stoneflies

These belong to the order Plecoptera, the nymphs of this group live in the benthic zone of clean streams and lakes. The nymphs have long antennae with a flat body and widely separated legs. Adult stonefly have long antennae and veined front wings, often dark grey in colour. The nymph does not lose its long tail parts when it becomes an adult. All species of Plecoptera are intolerant of water pollution (William, 2005) and are considered universal indicators of good quality water (Mason, 2002b). Two families, Perlodidae and Leutridae, were identified in the samples. Perlodidae nymphs are distinctive often having contrasting patterns of light and dark coloration as well having un-branched gills, while the Leuctridae are evenly brown or yellow-brown with pigmentation on the head (Fig 2.3). The body mass for Perlodidae recorded is 0.1-1mg while the range for Leuctridae was 0.1-1.5mg.

Perlodidae (5mm) Leuctridae(5mm)

Figure 2.3 Stonefly families found in this study

(Source: http://en.wikipedia.org/wiki/Plecoptera)

2.4.3 Coleoptera (Beetles)

Beetles are insects belonging to order Coleoptera, which has the largest number of known insect species (James and Joseph, 2003). Beetles feed on plants, fungi and invertebrates and are generally characterized by a hard exoskeleton and a hard forewing. Beetle larvae are always very different from adults in both forms and habits. They usually have only chewing and piercing mouth parts while adults develop biting mouth parts. Wings develop internally and do not appear until the pupal stage. The beetle larvae and adults do not compete with each other because they live in different habitats and eat different food.

Four families of beetles were identified, but for two families (Haliplidae and Gyrinidae), both larval and adult stage have been distinguished. The larvae of Gyrinidae resemble small centipedes having three pair of legs and one pair of gills at each abdominal segment except the last segment which has four hooks. The adult whirligig beetles (Gyrinidae) have the second and third pair of legs, modified into paddle blades. The body mass range of Gyrinidae was 1-3mg. The tails of Haliplidae larvae are never segmented and the length of the tail is less than one third of total head and body length. The head is round or pointed and the bases of the hind limbs are obscured by large plates. The family Haliplidae have a dorsally convex body. The colour is reddish or brownish. The body mass recorded for the Haliplidae was 1-4mg. The diving beetles (Dytiscidae) have sharp mandibles and are dark brown in colour. The underside of the body is not covered with short hairs. The body mass for the family Dytiscidae was 1-3mg

The Elminthidae are more or less cosmopolitan in distribution, occurring in appropriate fresh water habitat (Lawrence *et al.*, 2000). The body is slightly flattened, bristly with neither scale nor scale like setae. Hind wings are well developed, tarsi without segment and eyes are not strongly protuberant. The body mass range is 2-3mg for Elminthidae in this study.

Gyrinidae (4mm) Haliplidae (Adult and larvae 3 mm) Dytiscida (3mm)

Elminthida(4mm)

Figure 2.4 Beetle families.

(Source:http://www.google.co.uk/images?hl=en&q=Family%20dytiscidae%2C%20Haliplidae%20and%20gyrinidae%20%5D&um=1&ie=UTF-8&source=og&sa=N&tab=wi)

2.4.4 Hemiptera

The order of insects mostly known as true bugs. The animal’s mouth parts are distinctive, forming a beak for piercing and sucking out the liquid. The antennae in the Hemiptera are long and segmented and the tarsi of the legs are also segmented. Hemipterans do not undergo complete metamorphosis between the larval and adult phase, their nymphs resemble the adults to a large degree, and the difference between young and adult is the development of functional wings.

Hydrometridae. These animals have an elongated body with the head four times thinner than wide, and the swollen eyes are located behind the middle of the head. The antennae of these animals are positioned at the end of the head (Gooderham and Tsyrlin, 2002). The body mass range is 1-1.5mg

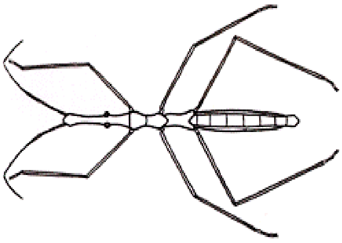
Corixidae. This animal is commonly known as the water boatman. The animal has a long flattened body and dark markings on the wings. They are strong fliers and are abundant in the later part of summer (Mau *et al.*, 2000). Body mass was around 1mg

Notonectidae. Also called back swimmers, they can be differentiated from Corixidae by their coloration and body shape. Hind legs are twice as long as the other legs. The mass ranged from 0.5 to 4mg in this study

Veliidae. These are commonly known as water crickets. The segments behind the head are wider than the rest of the abdomen, having a dark colour. They are found in streams, lake margins and sometimes away from water (Epler, 2006). The body mass range recorded was 0.02mg

Mesoveliidae. These are found on the surface of water and feed on small organisms found on the surface (Richard *et al.*, 1998). The insect is small, slender and yellowish in colour. The anterior wing is thickened and the antennae are long and slender. The body mass range was 1-2mg.

True bug (3mm)



Hydrometridae (5mm)

Cordixidae (4mm)

Notonectidae (5mm) Vellidae (3mm) Mesoveliida (4mm)

Figure 2.5 Six families of Hemiptera found in the samples

(Source:http://www.google.co.uk/images?hl=en&q=Hemiptera&um=1&ie=UTF-8&source=og&sa=N&tab=wi)

3.4.5 Neuroptera

Hemerobiidae. These are commonly known as the brown lacewing which has a dark spot on the wings. Hemerobiidae eggs are extremely cold tolerant (Garland, 1981) hatch in about 11 days, depending on the temperature (Klimaszewski and Kevan, 1985). The larvae form a pupa after third instar, which last about 9-14 days in the summer. The body size is small to medium in the range 1-4 mg and the mouth parts are modified to form long slender tubes.

Family Hemerobiidae (5mm)

Figure 2.6. Only one family of the order Neuropteran was found

(Source: http://www.google.co.uk/search?hl=en&q=Hemerobiidae&um=1&resnum=1&ie=UTF-8&sa=N&tab=iw)

3.4.6 Trichoptera (Caddisflies)

This aquatic larva is found in streams, lakes and ponds. The larvae make an open ended cylindrical case to hold the body but the head and legs remain outside the case. The legs are used for swimming and walking. The head capsule is well developed with chewing mouthparts. Thread-like abdominal gills are usually present in case-makers. In adults, the mouthparts are reduced or vestigial with two pair of wings clothed with long hairs. The wings are held over the abdomen and there are filiform antennae. The body mass range of the larvae was 0.5-3mg.

Rhyacophilidae. These have a greenish brown colour, and lateral abdominal gills. The animal is free swimming, is not case or net forming with the terminal abdominal prolegs (Watson and Dallwitz, 2003). The body mass range was 0.2-1mg.

Hydropsychidae. This animal is present in a wide range of river and streams habitats and has been chosen as an indicator species (Vuori, 1996). The larvae have tufted gills present underneath the abdomen. The body mass determined in this study was 1-5mg.

Polycentropidae. The animals lack abdominal gills, having hook shaped anal claws. Head usually conspicuously spotted and body often pink in colour. They live in a variety of aquatic habitat and found in Lake bottom in this study. Large number of Polycentropidae found in acidic stream during the study of post mining stream in lower Lusatia, Germany (Hünken and Mutz, 2007). The body mass range for this family was 0.1-3mg.

Caddisfly (5mm)

Rhyacophilidae (6mm)

Hydropsychidae (8mm)

Polycentropidae (7mm)

Figure 2.7 Order Trichoptera families recorded in this study

(Source: http://www.google.co.uk/images?hl=en&q=Trichoptera&um=1&ie=UTF-8&source=og&sa=N&tab=wi)

2.4.7 Diptera

The order diptera comprises the true flies, which have a simple four stage of life cycle. The larvae hatch from eggs and feed on available food until ready to enter the pupal stage. They metamorphose within their pupae into adult flies. The larvae of this group have no true legs but on some segments there are small prolegs. The larvae have distinctive mouth parts which appear hook like with gills at the hind end. The body mass range recoded was 0.5-4mg.

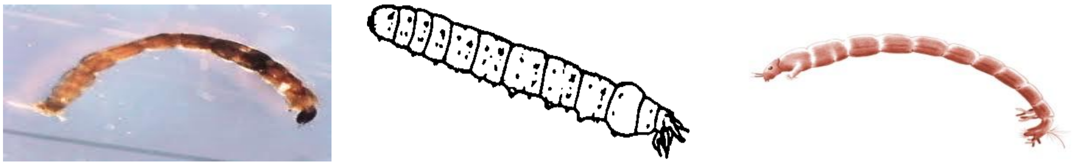
Tipulidae. Crane fly larvae which are worm like, thick skinned and brownish green to transparent in colour. There are six lobed plates at the rear end or two to six short tails and a lack of spines on the abdominal segments (McDonald *et al.*, 1990). The adults have a thin body shape, with narrow wings and long legs. The body mass for Tipulidae recorded here was 4-6mg.

Chironomidae (nonbiting midge larvae). The larvae can found in a very diverse range of freshwater habitats, have a prolonged front end, head horizontally held and are bright red in colour. Adult chironomidae resemble mosquitoes but do not bite. The body mass determined here was 0.01-0.4 mg for larvae.

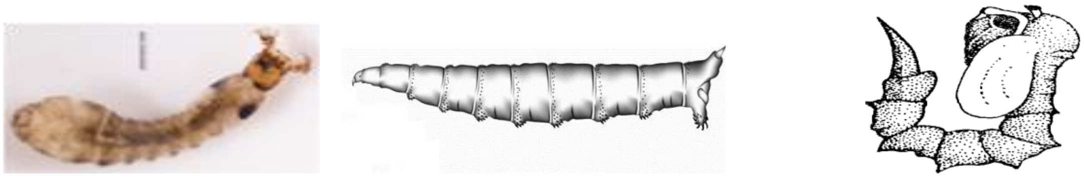
Simuliidae. The larvae are usually attached to the substrate and found in running water. The posterior end of the abdomen is noticeably swollen. Adult simulidae occur in various shades of grey or yellow, have a thorax shiny, are strongly convex, giving a humpbacked, gnat-like appearance. They have a small head with large round eyes and short 11-segmented antennae. The body mass recorded here was 0.02-2mg.

Ceratopogonidae. These are closely related to the Chironomidae, look like worms and are found in any aquatic habitat. The head capsule is usually complete and exposed, there is a lack of the prolegs on body segments, and the body segments and head are twice as long as broad. Adult females suck blood from other insects and mammals and all males feed only on nectar The body mass range was 0.05-2mg in this study.

Dixidae. The animal is of small size and is bent into a definite Ushape position in rest at the water meniscus. The body is yellowish to brown in colour and the wings are without scales. The head possesses mouth brushes and simple antennae. The larvae are found in running water. Adult are mosquito-like. Mouthparts are not developed into a sucking proboscis. Dixidae body mass range was 0.6-1mg in this study.



Ceratopogonidae(5mm) Tipulidae (8mm) Chironomidae(7mm)



Simulidae(5mm) Diptera larvae (5mm) Dixidae(3mm)

Figure 2.8 The larvae and nymph stages of true flies recorded here.

The Dixidae image coped from article (Redfern, 1975)

(Source:http://www.google.co.uk/images?q=Diptera&um=1&hl=en&ndsp=18&ie=UTF8&source=univ&ei=HX80TN7wIcvbsAbG1qTOBA&sa=X&oi=image_result_group&ct=title&resnum=5&ved=0CD0QsAQwBA)

2.4.8 Molluscs

Two groups of molluscs have been found in this research. The Gastropoda, include snails and the Bivalvia, or clams and mussels. The gastropoda have either a spiraled or coiled or low conelike shell. They are benthic organisms that slowly move about on the substrate of almost all shallow freshwater habitats. Calcium carbonate is used in the production of the shell, and it is for this reason that many freshwater snails are more common in hard water habitats. Mussels occur on or in the substrate. They generally feed by filtering planktonic microorganisms out of the water, although burrowing forms feed on organic detritus strained from the substrate.

Gastropoda. Eight families of Gastropoda were recorded in this research. The shell of family **Valvatidae** may be spiral and thick and of a dark green colour. The Valvatidae body mass recorded 0.9mg. The duck mussel **(Unionidae)** often with wings and with no clear depression between the umbo and blunt. The body mass determined here was 0.9-1mg for this family. The family **Planorbiidae** are coil shaped while family **Viviparidae** have a banded shell. The body mass range was 0.7-3mg for this family. The **Ancylidae** have a limpet shape, the spire arches backward and is inclined to the right. The body mass range was determined as 0.006-0.01mg for the family Ancylidae. **Hydrobiidae** are never banded and have an open umbilicus. The thickness of Hydrobiidae varies from thin to fairly thick. The body mass determined for this family was 0.3-0.4 mg.

The common name of the family **Physidae** is bladder snail. The shell is fragile and shows left handed coiling. The Physical body mass range was 0.004-0.006mg in this study. The family **Lymnaeidae** prefer still water and found in lakes and ponds. The shell is thin and medium to large size usually with pointed spire. The family Lymnaeidae are considered most tolerant to the water pollutants (Bogatov and Bogatova, 2009). The body mass range obtained here was 0.007-0.0085mg for the family Lymnaeidae.

Bivalvia. These animals have a shell consisting of two asymmetrically rounded valves which are a mirror image of each other and joined by a ligamentous hinge at one edge. The shell is held shut by strong muscles. The family Sphaeriidae were found in the lake and canal samples. The pea mussel **(Sphaeriidae)** is rounded and slightly oval shaped. The umbo is located at the centre and is slightly elevated above the hinged line, and the posterior end of the shell is longer than broad. The body range was 0.01-0.05mg.

Valvitidaedae(1mm)

Unionidae (1mm)

Planorbiidae(1mm)

Ancylidae(1mm)

Hydrobiidae (3mm) Viviparidae(2mm) Physidae(5mm) Sphaeriidae(3mm)

Lymnaeidae(4mm)

Figure 2.9 The groups of molluscs (snails and bivalves) recorded here.

The family Sphaeriidae is found only in lakes and canals.

(Source:http://www.google.co.uk/images?um=1&hl=en&tbs=isch%3A1&sa=1&q=Molluscs&aq=f&aqi=g2g-m2&aql=&oq=&gs_rfai)

2.4.9 Arachinida

Arachinida are jointed legged invertebrates which are easily distinguished from insects. Most Arachinids are terrestrial but some inhabit freshwater environments. The order Acarina includes mites and ticks. Acarina lack a visible body division and the abdomen has fused with the presoma. There are four pairs of legs, except for larval mites which possess three pairs of legs. The body mass ranged from 4.8 to 9.4 x 10^{-4} mg for Acarinida.

Acarina(1mm)

Figure 2.10 The Acarina recorded in the study.

(Source:http://www.google.co.uk/images?um=1&hl=en&tbs=isch%3A1&sa=1&q=Acariina&aq=f&aqi=g10&aql=&oq=&gs_rfai)

2.4.10 Crustaceans

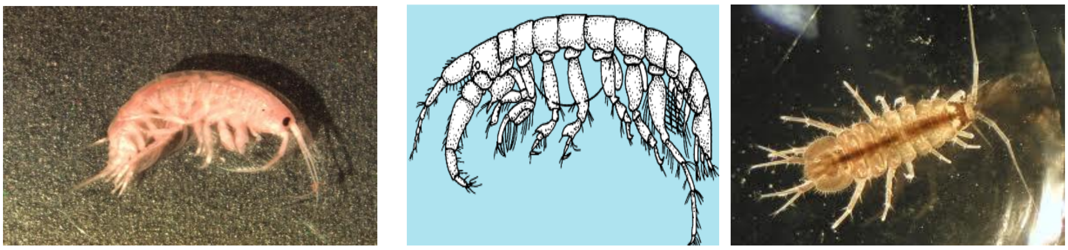
Crustaceans are group of arthropods which includes woodlice, shrimps, crabs and lobsters. Body segmentation is usually well developed. Body regions include a head and trunk or a head, thorax and abdomen. The head almost always bears two pairs of antennae. Most crustaceans are aquatic and found in marine or freshwater environments. Four groups of crustaceans were recorded here.

Amphipoda. Amphipods are found both in fresh and saline water (Wade, 2004). Juvenile amphipods look like the adults after hatching from eggs. Amphipods have a wide range of diets, depending on the species. Most live only for one year, reproducing once during their life time. Two families, Gammaridae and Corophiidae, have been identified here. Gammaridae are common invertebrates of streams and pools, having curved antennae, not more than half the length of the body. The body mass range was 0.3-3mg for the family Gammaridae. Corophiidae are found in pools and lowlands and identified by unique characteristics of their antennae, more than half as long as the body, often used in walking (Croft, 1986). The body mass range was 0.07-2mg for Corophiidae in this study.

Cladocera. These small crustaceans have the body enclosed by a kind of shell open at the bottom, which does not cover the head. The head and antennae are Y shaped, and are enlarged to form swimming organs. The animal is sensitive change in water quality (Sakamoto *et al.*, 2010). The Cladocera body mass recorded here was in the range 0.005-0.007mg

Asellidae. The body is segmented and flattened with six pairs of legs and two pairs of antennae. These are omnivores; generally feeding on decomposing plants. Juveniles tend to feed on faecal detritus and are predated by flatworms, fish and waterfowl. The first pair of antennae is long. The colour is brown and the legs are free of segments above, projecting well away from the body. The body mass range was 0.3-3mg for the family Asellidae.

Argulidae (Fish louse). The shape of this animal is oval and there are two suckers under the thorax which are clearly seen. Fish lice are free swimming and can survive several weeks without a suitable host. When the eggs hatch, the juvenile passes through several metamorphic changes as it develop into an adult. The animal has more than four pairs of legs. The body mass was estimated at 0.0015 mg.



Gammaridae (5mm) Corophidae (4mm) Asellidae(8mm)

Cladocera (5mm) Argulidae(3mm)

Figure 2. 11 Crustaceans classes found in moving and static water habitat

(Source:http://www.google.co.uk/images?um=1&hl=en&tbs=isch:1&&sa=X&ei=xJ80TJ3zIuSVOKre8KYE&ved=0CCMQBSgA&q=shrimp,+cladocera+argulus&spell=1)

2.4.11 Worms

This mixed group includes the annelids (leeches and earthworms) and flatworms (turbellaria). Worms are hermaphrodites, the juveniles hatching from cocoons, each cocoon holding one to five worms. It does not take long for the worm to grow into a full adult.

Nematomorpha. The Nematomopha are known as horse hair or Gordian worms and are long, thin and black in colour with body mass in the range 0.002-0.009mg. They are often wound around other objects or each other. The larvae hatch from the eggs and swim to an aquatic arthropod. They penetrate the body wall of the host by proboscis. When they mature, the worm leaves the host.

Turbellaria. These are members of the platyhelminthes. The body is elongated, relatively soft and lance-shaped with an earlike structure on each side of the head. After several days of nourishment inside the cocoon, turbellarians hatch from their eggs and released from their parents. Body mass was 1.3mg in this study. Freshwater turbellarians are normally freeliving and are found in streams, ponds, lakes and ditches.

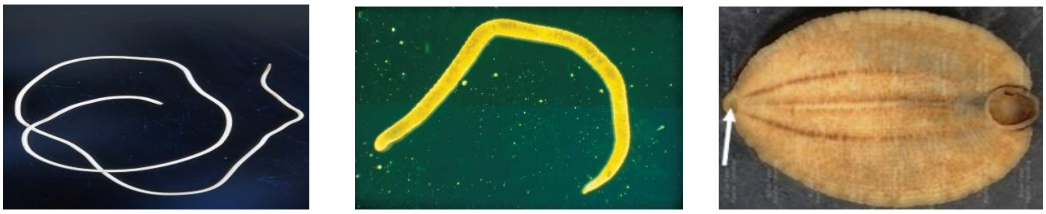
Leeches Leeches belongs to the Phylum Annelida. Silty habitats are unsuitable for leeches because they cannot tolerate high turbidity, but some species tolerate mild pollution (Mackie, 2001). Three families of leeches (**Glossiphonidae, Erpobdellidae and Piscicolidae**) have been identified here. Predacious leeches are found attached to submerged objects such as stones, trees or aquatic vegetation. Leeches deposit their eggs inside cocoons. The juvenile emerges from the cocoons and achieve adulthood when they reach their critical body weight or once they reach sexual maturity. Body mass found in range of 0.4 -131mg for these families.

Oligochaeta. These are mainly terrestrial and freshwater animals with markedly segmented body. The body segments have small number of bristles but have no other appendages and the body mass range in this study was 0.1-3mg. The earthworms cocoons deposited in the soil. On hatching, the young worms resemble small adults and grow continually until they reach maturity.

Glossiphonidae. These are the member of Rhynchobdellida with flattened body with fused eyespots. In freshwater systems, species that feed from vertebrates are out-numbered both in number of species and absolute abundance by predatory leeches (Klemm, 1972; Sawyer, 1986). Glossiphonidae serve as environmental stress indicators due to their relative abundances in freshwater habitats (Grantham and Hann, 1994). Glossiphonidae body mass ranged from 16 to 52 mg which is at the larger end of the size range of benthic mass distributions in the present study.

Piscicolidae. These have cylindrical bodies with two bellshaped suckers on both side of the body. Some Piscicolidae have less specificity and detach from their host during their life cycle. Most Piscicolidae leeches feed on several hosts and do not remain attached after each feeding session but reattach after following digestion of the blood meal (Sawyer and Hammond, 1973). The body mass range was 1-4mg for this family.

Erpobdellidae. These are one of the most common freshwater leeches (Soos, 1968). They have four pairs of eyes arranged horizontally, but never have true jaws. Erpobdellidae leeches feed on small invertebrates (Smith, 2001) and are found with high densities often related to organically polluted running aquatic habitats (Matyziak, 1979). The largest body mass range of 6- 140 mg was obtained for this family.



Oligochaeta(8mm) Nematomorpha(8mm) Glossiphonidae (5mm)

Erpobdellidae (9mm) Piscicolidae (6mm) Turbellaria(5mm)

Figure 2.12 Groups of worms recorded in freshwater streams and lakes

(Source:http://www.google.co.uk/images?q=Worms+leeches+and+turbellaria&um=1&hl=en&tbs=isch:1&ei=drE0TNG5MsLgOOuipIYM&sa=N&start=54&ndsp=18)

2.5 Body Size Determination

Body size was mainly determined by measuring a body dimension (length or head width) under a calibrated low power microscope fitted with a graticule eyepiece. The body lengths of animals were determined as the distance from the anterior of the head to the end of the last abdominal segment (cerci and other appendages were excluded). Head capsule width was always measured across the widest section of the head. Previously established relationships (allometric equations) from the literature (Stead *et al.*, 2003; Feller and Warwick, 1988; Leaper *et al.*, 2001; Smock, 1980; Towers *et al.*, 1994), approximate 2-D or 3-D geometric shape and water displacement techniques were all applied as appropriate to derive the dry body mass of all the taxonomic groups. The water displacement method was used for larger individuals, by placing within a standard millilitre syringe. The water level in the syringe was read, the animals completely submerged and the level read again after 30s. The difference between the two readings was recorded as volume in microlitres (µL), converted to mass (µg) multiplying by 1.05 then to mg. The regression equations (allometric relationships) used for different groups of invertebrates are shown in Table 2.1 below.

Table 2.1: Regression equations to determine body mass of invertebrates communities where DM is a dry mass of the organisms (mg), DW is the dry weight (mg) of the organisms, L is the length (mm) of the organisms, HW is the head width (mm) of the organisms, volume (V) of the organisms expressed in (nL) and Wt is weight of the organisms (mg).

Family/ Class	**Regression Equation**	**Reference**
Baetidae Heptageniidae Caenidae Ephemerellidae Ephemeridae Potamanthidae Hymenoptera	$Dw(mg)= aL(mm)^{b}$ $Dw(mg)=3.8x10^{-3}L(mm)^{2.918}$	(Stead *et al.*, 2003)
Leuctridae Perlolidae	$Dw(mg)=aL(mm)^{b}$ $DW(mg)=2.5x10^{-3}L(mm)^{2.744}$	
Gyrinida Dyticidae Haliplidae Elminthidae	$Dw(mg)=Ina+bInL(BL(mm)or\ HW(mm)$ $Dw=-2.0076 +3.2271\ InL(BL-Dw)$ $Dw =3.1102 +2.5412\ InL(HW-DW$	(Towers *et al.*, 1994)
Corixidae Notonectidae Mesovellidae Veliidae Hydrometridae Hemiptera	$In\ W(mg) = Ina + b\ In\ L$ $In\ W(mg) = -4.200+ 2.60In\ L(mm)$	(Smock, 1980)
Hemerobiidae	$Log\ DM(\mu g)=a + b\ log\ HW(mm)$ $Log\ DM(\mu g)=2.68+2.9\ log\ Hw(mm)$	(Stead *et al.*, 2003)
Trichoptera Rhyacophilidae Hydropsychidae Polycentropodidae	$In\ DM(mg)=Ina + b\ InL(mm)$ $In\ DM\ (mg) = -6.037 + 2.82\ In\ L\ (mm)$	
Tipulida	$DW(mg) = aL(mm)^{2.851}$ $Dw(mg)= 1.3x10^{-3}\ L(mm)^{2.851}$	

Simulidae	In DM(mg)=Ina + b Hw(mm) In DM(mg)=-4.5009 +2.0742 HW(mm	
Chironomidae Ceratopogonidae	DM (mg) = a L(mm)^{b} DM (mg) = 6.0x 10^{-4} L (mm) $^{2.770}$	
Dixidae	DM(mg)=aL (µm)^{b} DM(mg)=6.62x10^{-4} L(µm)$^{2.59}$	
Valvatidae Uniondae Planorbidae	**Water Displacement** V(nL)=WD(µL)x 1000 Wt(µg)=v(nLx1.05) Mass(mg)=µg/1000	(Leaper *et al.*, 2001)
Hydrobiidae Physidae Viviparidae Lymnaeidae	V(µL) = L(mm) (0.851)$^{1.91}$ Wt(µg) = v(µLx 1.05) Mass(mg) = µg/1000	
Ancylidae Sphaeriidae	**Approximate a geometric shape(cone)** V(µL) = 1/3πr^{2} (mm) h(mm) V(nL)= µLx1000 Wt(µg)=nLx1.05 (mg) = µg/1000	
Arachinida Argulidae	DM(µg) = aL(µm)^{b} DM(µg) = 1.1x10^{-5}L(µm)$^{1.89}$	(Stead *et al.*, 2003)
Gammaridae Corophidae	In DM(mg)=Ina +b InL(mm) In DM(mg)=-4.95 +2.83 InL(mm)	
Asellidae	DM(mg) = aL(mm)^{b} DM(mg) = 7.2x10^{-3}L(mm) $^{2.785}$	
Cladocera	InDM(µg) = Ina+bInL(mm) InDM(µg) = In1.7512+ 2.653L(mm	
Oligochaeta	DM(nl) =a L(µm)^{b} DM(nl) = 3.5 x10^{-3} L(µm)$^{2.1}$	
Nematomorpha	DM(µg)=a L(µm)^{b} DM(µg) =6.0x10^{-5} L(µm)$^{0.8205}$	
Turbullaria	V(nL)=L(mm)xW^{2}mm) x C V(nL)=L(mm)W^{2}(mm)x 550 V(nL) ×1.05 = dry weight= *µg* *µg/1000=mg*	(Feller and Warwick, 1988)
Piscicolidae Erpobdellidae Glossiphonidae	V(nL) = L(mm)×π(W/2)^{2} ×530 V(nL)×1.13 = dry wight(µg)	

2.6 References

Bass, J. (1998). *Last instar larvae and pupae of the simulidae of Britain and Ireland: A key with brief ecological notes.* Institute of Freshwater ecology.

Bogatov, V. and Bogatova, L. (2009). Heavy metal accumulation by freshwater hydrobionts in a mining area in the south of the russian far east. *Russian Journal of Ecology* 40(3): 187-193.

Croft, P. S. (1986). *A key to the major groups of British freshwater invertebrates.* FSC (Field Studies Council)

Elliott, J. M. H., U. H. and Macan, T. T (1988). *Larvae of the British Ephemeroptera: A key with ecological notes.*: *Scientific Publications of the Freshwater Biological Association*.

Epler, J. H. (2006). *Identification manual for the aquatic and semi-aquatic heteroptera of Florida.*: Division of Water Resources Management, Department of Environmental Protection, State of Florida, Tallahassee, FL.

Feller, R. J. and Warwick, R. M. (1988). *Energetics. In: Introduction to the study of meiofauna. Smithsonian institution press, Washington,.* Smithsonian Institution, Washington DC.

Gooderham, J. and Tsyrlin, E. (2002). *The waterbug book: A guide to the freshwater macroinvertebrates of temperate Australia.* CSIRO Publishing.

Grantham, B. A. and Hann, B. J. (1994). Leeches (annelida: Hirudinea) in the experimental lakes area, Northwestern Ontario, Canada: Patterns of species composition in relation to environment. Canadian. *Journal of Fisheries and Aquatic Sciences* 5: 1600-1607.

Hünken, A. and Mutz, M. (2007). On the ecology of the filter-feeding neureclipsis bimaculata (Trichoptera, Polycentropodidae) in an acid and iron rich post-mining stream. *Hydrobiologia* 592(1): 135-150.

James, K. L. and Joseph, V. M. (2003). In *Encyclopedia of Insects* (Eds. Resh, V. H and Crade, R.T). Academic press.

Klemm, D. J. (1972). The leeches (Annelida: Hirudinea) of Michigan. *Michigan Academician* 4: 405-444.

Lawrence, J. F., Hastings, A. M., Dallwitz, M. J., Paine, T. A. and Zurcher, E. J. (2000). Elateriformia (Coleoptera): Descriptions, illustrations, identification, and information retrieve for families and subfamilies.

Leaper, R., Raffaelli, D., Emes, C. and Manly, B. (2001). Constraints on body-size distributions: An experimental test of the habitat architecture hypothesis. *Journal of Animal Ecology* 70(2): 248-259.

Mackie, G. L. (2001). *Applied aquatic ecosystem concepts.* Kendall/Hunt Publishing Company.

Mason, C. F. (2002). *Biology of fresh water pollution.*: Pearson Education Limited.

Matyziak, K. (1979). Structure of leech groups (Hirudinea) in polluted parts of the catchment area of the Rivers Bzura and ner.Part 1. Field investigations. *Acta Hydrobiologica (Krakow)* 18: 259-276.

Mau, H. E. L., Foottit, R. G., Hamilton, K. G. A. and Scudder, G. G. E. (2000). Checklist of the Hemiptera of Canada and Alaska. NRC *(National Research Council) Research Press, Ottawa.*

McDonald, B., W. Borden and Lathrop, J. (1990). Citizen stream monitoring: A manual for Illinois. *Illinois Department of Energy and Natural Resources,* ILENR/RE-WR-90/18. Springfield, Illinois.

Munn, M. and King, R. (1987). Ecology of (Walsh) Potamanthus myops (Ephemeroptera: Potamanthidae) in a Michigan stream (USA) *journal Hydrobiologia* 146: 71-75.

Peter, H. and Matthew, Y. (2001). On the calibration of silverman's test for multimodality. *Statistica Sinica* 11: 515-536.

Prazen, E. (1962). On the estimation of a probability density and mode. *Annals of Mathematics and Statistics* 33: 1065-1076.

Redfern, M. (1975). Revised field key to the invertebrate fauna of the stony hills streams *Field Studies* 4: 105-115.

Richard, E., White, D. J. B. and Peterson, R. T. (1998). *A field guide to insects.* Houghton Mifflin Co.

Rosenblatt, M. (1956). Remarks on some nonparametric estimates of a density function. *The Annals of Mathematical Statistics* 27(3): 832-837.

Sakamoto, M., Ogamino, Y. and Tanaka, Y. (2010). Leptodora kindtii: A cladoceran species highly sensitive to toxic chemicals. *Limnology* 11(2): 193-196.

Sawyer, R. T. (1986). *Leech biology and behavior.* Oxford University Press, Oxford, U.K.

Sawyer, R. T. and Hammond, D. L. (1973). Observations on themarine leech Calliobdella carolinensis (Hirudinea: Piscicolidae), epizootic on the atlantic menhaden. *Biological Bulletin. Marine Biological Laboratory,Woods Hole* 145: 373-388.

Seaman, D. E. and Powell, R. A. (1996). An evaluation of the accuracy of kernel density estimators for home range analysis. *Ecology* 77(7): 2075-2085.

Silverman, B. W. (1981). Using kernel density estimates to investigate multimodality. *Journal of the Royal Statistical Society. Series B (Methodological)* 43(1): 97-99.

Smith, D. G. (2001). *Annelida. Pennak's freshwater invertebrates of the United States: Porifera to crustacea.* John Wiley & Sons.

Smock, L. A. (1980). Relationships between body size and biomass of aquatic insects. *Freshwater Biology* 10(4): 375-383.

Soos, A. (1968). Identification key to the species of the genus Erpobdelladae Blainville1, 818 (Hirudinoidea:Erpobdellidae). *Annales Historico-Naturales Musei Nationalis Hungarici* 60: 141-145.

Stead, T. K., Schmid-Araya, J. M. and Hildrew, A. G. (2003). All creatures great and small: Patterns in the stream benthos across a wide range of metazoan body size. *Freshwater Biology* 48(3): 532-547.

Towers, D. J., Henderson, I. M. and Veltman, C. J. (1994). Predicting dry weight of New Zealand aquatic macroinvertebrates from linear dimensions. *New Zealand Journal of Marine and Freshwater Research* 28(2): 159 - 166.

Vuori, K. M., Parkko, M. (1996). Assessing pollution of the River Kymijoki via hydropsychid caddis flies. *Archiv für Hydrobiologie* 136(2): 171-190.

Wade, S. Tracy, C and Linda-Marie, M. (2004). Critter catalogue. A guide to the aquatic invertebrates of south Australian Inland waters. Waterwatch South Australia.

Watson, L. and Dallwitz, M. J. (2003). British insects: The families of Trichoptera. Version: 28th august 2009. Insect groups treated separately in the 'British insects' package.

Whittle, P. (1958). On the smoothing of probability density functions. *Journal of the Royal Statistical Society. Series B (Methodological)* 20(2): 334-343.

William, H. R. (2005). *Handbook of urban insects and Arachnids.* Cambridge University Press.

Chapter 3

The body size of macro-invertebrates in the Rivers Ure and Foss: effects of water quality, season and sampling methodology

3.1 Introduction

The purpose of this chapter is to evaluate general water quality, using the resident biological organisms (macro-invertebrates) and then to explore the distributions of body masses of stream invertebrates in relation to that water quality. Holling (1992) argued that ecological processes in undisturbed ecosystems are discontinuously distributed and species functioning at different scales respond differently to the opportunities provided at each of these scales resulting in discontinuities in processes. Furthermore, these processes are closely related to their body mass patterns because body mass is an indicator of a wide range of processes (see chapter 1). Such distributions have now been reported in a variety of habitats (Havlicek and Carpenter, 2001; Kamenir *et al.*, 2004; Bakker and Kelt, 2000). Body mass distributions are expected to be multimodal in undisturbed natural communities and disturbed communities we might expect in the number of modes to decline as disturbance increases.

Detecting multimodality is challenging. Holling (1992), in exploring his Textural Discontinuity Hypothesis, developed the Body Mass Difference Index (BMDI) to detect discontinuities (gaps) in body mass patterns. Others have used Kernel Density Estimation (KDE) to estimate the number of modes, and by default, the number of troughs in distributions e.g. (Raffaelli *et al.*, 2000; Havlicek and Carpenter, 2001). Silverman's method for KDE is considered conservative in estimating the actual number of modes, even in large samples (Hall and York, 2001), so that in this chapter, both BMDI and KDE were used to identify the degree of lumpiness in body mass distributions.

The data sets used here to explore discontinuities in body mass distributions were from freshwater stream communities in Yorkshire, specifically the River Ure and the River Foss. The sites were chosen on the basis of the range of water qualities they potentially offer. Seasonal variations in community composition and its effect on the number of modes were also investigated for one site, Wensley on the River Ure. The differences in species composition and body mass patterns which could be obtained by using different sampling approaches (Kick and Surber sampler), are explored here for another site, Tang Beck in York, part of the River Foss system.

Specifically, this chapter addresses the following questions:

1. Are the freshwater benthic assemblages at these sites "lumpy" in their body mass distributions, that is, multimodal or having gaps in their distributions?

2. Is there a relationship between water quality (disturbance) and the degree of lumpiness, as reflected in the number of modes and /or the number of gaps?

3. Are 1 and 2, above, influenced by season and /or choice of sampling technique?

3.2 Sampling Sites

3.2.1 River Ure and its tributaries

The River Ure rises in the Pennines within the Yorkshire Dale National Park, with several small becks combining around the town of Hawes to form the main river. The river is further joined by other tributaries like the River Burn just south of Masham, the River Skell to the east of Ripon and the River Tutt at Borough Bridge. The Ure, Swale, and Ouse combined catchment area covers about 3200km^2 (Figure 3.1). After merging with the River Swale, the River Ure flows south east. The water quality of the higher altitude stream is good, but it becomes increasingly polluted downstream due to urban drainage and sewage discharges (Anonymous, 1997). The landscape through which the river flows spreads from the Pennines in the north and west, and is mainly rural but has industry in some villages and towns in the south. Several major roads cross the area, including the A1, A19 and A164. The soil in the area supports agriculture whilst sandstone and carboniferous Millstone Grit form areas of grit moorland. Millstone Grit is considered a major aquifer and is generally a source of good quality water. These rocks occur in a variety of sizes from boulders to gravel, and sand to silt. Some non-aquifer rocks, which are important in preventing contamination to the underlying aquifers, are also present in the area. Three locations were sampled on the River Ure, and these are described below.

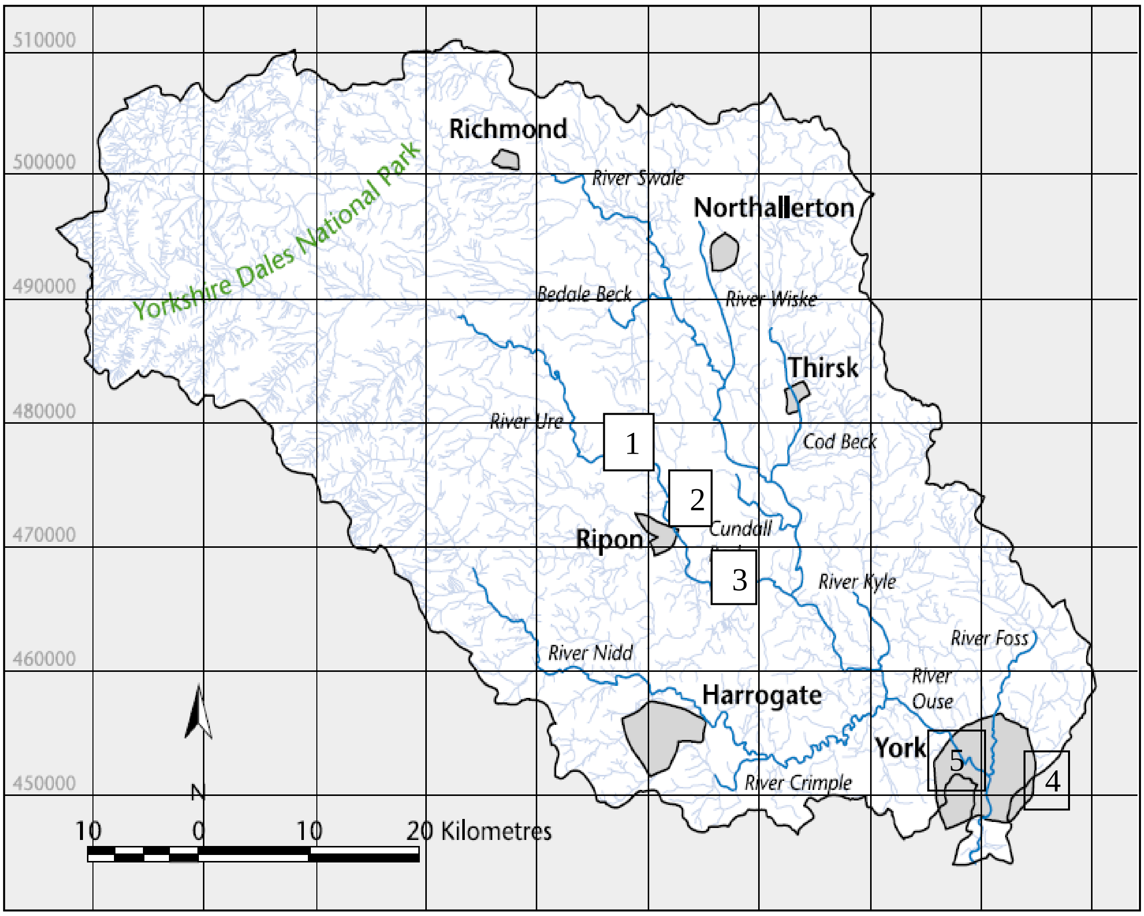


Figure 3.1. The Swale and Ure system. Main study sites are the River Ure at Wensley, Masham and Ripon indicated as 1, 2 and 3 respectively while River Foss = 4 and Tang Beck =5.

(Source:Anonymous)

3.2.2 River Ure at Wensley

Wensley is a small rural village located between the hills southwest of the town of Leyburn. The soils are fertile with tracts of cultivated ground, wood and pasture, making the scenery striking and beautiful. The River Ure at Wensley has a generally healthy fishery for angling and is popular for recreation (Anonymous, 1997). The water quality at Wensley is considered good. There is no industry nearby, reducing the chances of pollution.

Figure 3.2. River Ure at Wensley. The main sampling site is the shingle riffle at the left side of the bridge.

(Source: http://www.geograph.org.uk/photo/1435803)

3.2.3 River Ure at Masham

At Masham, almost 32 km downstream from Wensley, the river increases in size. Poor quality effluents enter the river at Masham in the form of domestic wastes (substantial amounts of ammonium and phosphorus) (Lewis *et al.*, 1997). Masham is famous for its local beers, which are made by the traditional method of brewing. The method was developed 200 years ago and black local ingredients and clear Dale water are still used to produce the distinctive Blacks Sheep brew. The brewery effluent with high organic contamination enters the river and reduces the quality of water at Masham (Littlewood *et al.*, 1998).

Figure 3.3. River Ure at Masham

(Source:http://www.google.co.uk/images?q=river+ure+near+masham&um=1&hl=en&tbs=isch:1&ei=tmdITKDGOIKUjAe2nK22Dg&sa=N&start=18&ndsp=18)

3.2.4 River Ure at Ripon

Ripon is a small town, 16 km further downstream from Masham. Here, the main river is joined by other tributaries below Masham, which increase the width of river so that at Ripon the river provides good coarse fishing. Gypsum occurs at relatively shallow depth in rocks at this site, which are susceptible to dissolution by circulating groundwater leading to subsidence (Cooper and Waltham, 1999). The tributaries introduce minor pollutants to the river which results in lower water quality at the site.

Figure 3.4. River Ure at Ripon

(Source:http://www.google.co.uk/images?hl=en&q=river%20ure%20at%20ripon&um=1&ie=UTF-8&source=og&sa=N&tab=wi)

3.2.5 Tang Beck

Tang Beck is at Haxby in York. The Tang Hall area is a remnant of old countryside in the town and the beck is a tributary of the River Foss. Storm drains overflow during heavy rainfall and the site is considered polluted due to domestic waste. Approx 2.5 hectares of wet grassland are situated alongside the beck which precludes bank side development because of regular winter flooding.

Figure 3.5. Haxby Tang beck

(Source:http://www.google.co.uk/images?q=Tang%20beck&um=1&hl=en&ndsp=18&ie=UTF-8&source=og&sa=N&tab=wi)

3.2.6 River Foss

This river is a tributary of the River Ouse. It rises at Oulston in Hambleton, 6.5km north of Easingwold. The river is 31.3 km in length (Anonymous, 2009) and flows through low-lying clay areas of fairly intensive mixed farming and small villages, and then through the highly developed urban areas of York. The samples were collected from River Foss near York. The river finally discharges into the River Ouse in the centre of York. As a result of domestic sewage discharges, the levels of nutrients are very high in the Foss and the area is categorized as eutrophic sensitive (Defra, 2010). Algal development and weed growth are established due to the slow flow and this depletes oxygen and leads to fish kills occasionally (Welker *et al.*, 2001).

Figure 3.6. River Foss at York.

(Source: http://www.google.co.uk/images?hl=en&q=river%20foss&um=1&ie=UTF-8&source=og&sa=N&tab=wi)

All the above sites were sampled for their benthic invertebrates using a Surber sampler or kick sampling net, as appropriate (see chapter 2). Wensley was sampled in both the spring and summer in order to explore the effect of season on body size patterns. All other sites were only sampled in the spring using the Surber technique. At Tang Beck, both the Surber sampler and the kick sampling net were used to see the effect of the sampling technique on body mass patterns. Tang Beck also has a consolidated clay bed which makes the use of the Surber sampler quite difficult.

3.3 Analysis of Water Quality at the Sites

3.3.1 Biotic Indices

A number of biotic indices are available to evaluate water quality (Metcalfe-Smith, 1996), including the Saprobic Index, Biological Monitoring Working Party Score (BMWPS), Index of Biotic Integrity (IBI), Macro-invertebrates Community Index (MCI) and Average Score Per Taxon (ASPT). For all of these indices, the sensitivity and tolerance of a given species or taxon to pollution are considered and a score or value assigned. The sum of the values for all the species or taxa in a sample gives a water quality index for that site. **Saprobien groups,** such as algae, protozoa, bacteria and rotifers, are used to determine the eutrophic state of the water. This method was designed specifically to detect only organic pollution, so that other pollutants, such as metals, diffuse runoff, etc, are poorly detected using this index.

The **Index of Biotic Integrity (IBI)** is an approach that compares the invertebrates found at a monitoring site to what might be expected, using a standard baseline condition that reflects little or no human impact (Karr, 1996). The IBI index uses a number of measurements to assess the biological condition, or health, of a stream and the ability of a system to support and maintain a balanced species composition. Fish species are normally used in this approach.

The **Macro-invertebrates Community Index (MCI)** is a qualitative method, where the absence or presence of indicator species within the sample is examined. MCI was developed by Stark (1993) in New Zealand for bio-monitoring in stony riffle. Absence of organisms is obviously important in assessing the effect of pollution (Azrina *et al.,* 2006) but it can make the interpretation of a pollution index very difficult due to seasonal changes in the abundance of animals if samples are taken at different times of year. The MCI uses a five-point scale of coded abundances (Absent, Rare, Common, Abundant, and Very Abundant). The MCI give little information about the status of the stream

The **Biological Monitoring Working party (BMWP)**, was set up in the United Kingdom in 1976 (ISO, 1979). The objective was to develop a system which would be suitable for the biological assessment of all rivers in the UK (Metcalfe, 1989). The method used the Chandler Score system at the starting point, to improve a standardized biotic system for assessing the quality of rivers (ISO, 1979). For BMWP, all groups are identified to the family level in order to ensure taxonomic uniformity. The abundance factor used in the Chandler Score was eliminated to create a simplified system introduced as the BMWP. In this system the score for each family is given between 1 and 10 (instead of 1-100) on the basis of their perceived susceptibility to pollution. The BMWP is the sum of the score of each family present in the sample. Values greater than 100 are associated with clean water, while less than 10 relate to very polluted rivers.

The **Average Score Per Taxon (ASPT)** is an average value of the sum of scores for each family of invertebrates present in a sample. The method was described by Hawkes (1998), and is widely used in UK rivers as it provides a quantitative method for assessing the condition of running water and gives a broad indication of biological condition (Hawkes, 1998). The method can be used for both kick net and Surber samples because the ASPT score is not dependent on sample size. Families with low tolerance to poor water quality are given high indicator values, whereas families with high tolerance are given low indicator values. The ASPT is a sum of the indicator values of all families, standardised (divided) by the number of taxa (families) present in a sample. A high ASPT indicates high ecological status and a low value reflects degraded conditions. ASPT declines with increasing pollution and hydromorphological stress, although changes in nutrient and temperature conditions might also change ASPT values. Pinder et al (1987), in their study on streams, compared the performance of different diversity and biotic indices at a single site and found the ASPT to be relatively independent of sample size, sample technique and season. Similarly, the ASPT and other techniques have been used in different rivers in the UK to evaluate water quality and better results are reported for ASPT (explaining 65% of the variance, as opposed to 22% for other results) (Armitage *et al.*, 1987). A significant dependence of the ASPT approach on temperature has been reported in a two year study of water quality (Zamora-Muñoz *et al.*, 1995).

General Stream Habitat

Water quality assessment using Biological Monitoring Working Party score (BMWP) and Average Score per Taxon (ASPT) need to be interpreted with caution due to possible effects of habitat structure. The Environment Agency use the River Invertebrates Predictions and Classification System (RIVPACS) to predict the expected score for a given habitat structure, This is then compared to the observed score for that habitat. In this way potentially confounding effects of stream habitat on the score are accounted for.

3.3.2 General Classification of Water Quality

There are six categories of general quality assessment (GQA) of water: very good, good, fairly good, fair, poor and bad. These classes have been determined by the Environment Agency in England, by combining two parameters, an ecological quality index based on the ASPT measure described above and the taxa present in the water body (Mason, 2002b). The computer model, RIVPACS (River Invertebrates Prediction and Classification System), has been developed to assess environmental stress based on the physical, geographical and chemical characteristics of a site, and what the invertebrate fauna of that site would look like in the absence of pollution. A comparison of the predicted macroinvertebrates communities with those actually observed allows calculation of ecological quality indices (EQI). The most relevant EQIs in describing biological quality are based on the number of macroinvertebrate taxa and ASPT as follows:

EQI taxa = Observed number of taxa present on given habitat (predicted from RIVPACS)

EQI ASPT = Observed ASPT for the present taxa on given habitat (Predicted from RIVPACS)

An EQI value of one indicates that the observed macroinvertebrate fauna is what would be expected in an unstressed or clean river, whereas a lower value of EQI indicates that communities are stressed to some degree. The descriptions of status of these six classes of ecological quality index (EQI Taxa-ASPT) are given in Table 3.1. An appropriate grade according to this system is given to the rivers recognised as poor or bad. The biology of the rivers-based EQI-ASPT may differ from site to site for intermediate quality in term of actual taxa present, but between the best and worst possible quality the grade reflects the relative position of the site on a scale of very good to bad.

Table 3.1 Lower limits of the biological grades for EQI-ASPT and EQI N-Taxa, system used in deriving the Biological General Quality Assessment (BGQA) for aquatic habitats.

Grade	EQI for ASPT	EQI for number of taxa	Environmental quality
A	1.00	0.85	Very good
B	0.90	0.70	Good
C	0.77	0.55	Fairly good
D	0.65	0.45	Fair
E	0.50	0.30	Poor
F	<0.50	<0.3	Bad

Grade a - Very Good

The biology is similar to or better than that expected for an average, unpolluted river of similar size, type and location. There is a high diversity of families, with several species in each and generally no dominance of a single family.

Grade b – Good

The biology shows minor differences from grade ‘a’ and falls a little low of that expected of an unpolluted river of this size, type and location. A small reduction in pollution sensitive families and a moderate increase in the number of individuals in the families that tolerate pollution like chironomids and worms. This may indicate the first signs of organic pollution.

Grade c - Fairly Good

The biology is worse than that expected for an unpolluted river of this size, type and location. Many of the sensitive families are absent or the number of individuals is reduced. In many cases there is a marked rise in the number of individuals in the families that tolerate pollution.

Grade d – Fair

The biology shows large differences from that expected for unpolluted river of this size, type and location. The site contains only a small number of pollution sensitive families and a large number of pollution tolerant families, with high numbers of individuals.

Grade e – Poor

Pollution tolerant families are dominant in terms of number of individuals. Sensitive species will be rare or absent.

Grade f – Bad

The biology is limited to a small number of very tolerant families, often only worms, midge larvae, leeches and the water hog-louse. These may be present in very high numbers but even these may be missing if the pollution is toxic. In the very worst condition there may be no life present in the river.

The procedure of biological assessment is simple enough to be used by a non-biologist after a short period of training and it allows the rapid investigation of sources of pollution with the minimum of resources (Rutt *et al.*, 1993), but there is average risk of 22% that rivers may be classified wrongly.

The sites sampled here were classified using these various approaches for their general quality conditions. The number of taxa, the abundance of sensitive and tolerant species and ASPT values for the sites were calculated and matched with this system. An Ecological Quality Index (EQI) for number of taxa could not be determined due to lack of EQI-N-taxa (predicted from RIVPACS) for the sites. Thus the sites were classified into different categories using biological indices.

3.3.3 Analysis of Body Size Distributions

The body sizes of all macroinvertebrates recorded in the samples were determined by measuring body dimensions and scaled to mass using allometric equations, as described in chapter 2. These data were then used to construct size frequency histograms of body size, as well as being analysed for the number of gaps in distributions using the BMDI approach and the number of modes by Kernel Density Estimates (KDE), as described in chapter 2.

3.4 Results

3.4.1 Invertebrate densities

A total of 34 taxa were identified in this study (Table 3.3). There were significantly fewer individuals in the River Foss samples relative to other three locations. The highest number of taxa (families) was present in the River Ure at Masham (Table 3.2), while the number of individuals is highest at Wensley. Community abundance (Figure 3.7) for the sites shows that large numbers of pollution sensitive taxa are present at Wensley and Masham. Wensley is dominated by Potamanthida, Chironomidae, Baetidae, Caenidae and Ephemeridae. At Ripon, there was a high abundance of stonefly and mayfly families (Perlodidae, Ephemerellidae, Leuctridae, and Baetidae). Most beetle larvae (Haliplidae), Simulidae, Ephemerellidae and Perlodidae were found at Masham. Chironomidae, Hydrobiidae and Gammaridae made up a large proportion of the taxa at Tang Beck, while the River Foss is dominated by Asellidae and Erpobdellidae. Tang Beck and the River Foss are dominated by pollution tolerant species, and a high abundance of Nematomorpha and Oligochaeta. At Tang Beck, 51% of individuals are Chironomidae, while 35% of individuals on the Foss are worms. On the Foss, intolerant taxa, such as Leucatridae, Perlodidae, Ephemerellidae and Baetidae, were absent and are less abundant in Tang Beck.

The community index (Table 3.2) for the sites shows that taxon richness is higher at Masham and Wensley. Indicator species EPT (Epheremeroptera, Plecoptera and Trichoptera) make a large proportion of individuals at Wensley, followed by Masham. The highest densities of tolerant invertebrates were recorded in the River Foss and the lowest are at Masham, while sensitive taxa make up a large proportion at Wensley and are fewest in Tang Beck and the River Foss, 6% and 7% respectively.

Table 3.2 Community Index of benthic communities for different stations on the River Ure, Tang Beck and the River Foss, collected at the same time (spring) and also in different seasons of year (at Wensley). The samples at Tang Beck were collected quantitatively and non-quantitatively. (S) indicates summer data while (SQ) indicates semi-quantitative.

Category	Wensley	Masham	Wensley(S)	Ripon	Tang	Tang(SQ)	Foss
Species Richness							
Total number of taxa	20	21	13	14	17	8	13
Number of EPT	6,2,2	6,2,1	2,2,1	2,2,1	1,0,0,	1,0,0	0,0,0
Number of Ephemeroptera	254	153	26	88	5	1	
Number of Plecoptera taxa	12	57	93	108	0	0	
Number of Trichoptera taxa	11	1	9	1	0	0	1
Composition Measures							
% EPT	56%	51%	27%	46%	3.2%	3%	7%
% Ephemeroptera	53%	37%	5.6%	32%	3.2%	1.3%	0%
Tolerance /Intolerance Measure							
Number of intolerance taxa	10	9	5	5	1	0	
% Intolerant taxa	50%	42%	38%	35%	5.8%	12%	7%
% tolerant taxa	20%	19%	30%	28%	52%	37%	61%

Table 3.3. Taxa recorded at sites on the River Ure, Tang Beck and the River Foss (+ present, - absent). S = summer and SQ = Semi-quantitative samples.

Taxa	**Wensley**	**Wensley(S)**	**Ripon**	**Masham**	**Tang**	**Tang(SQ)**	**Foss**
Chironomidae	+	+	+	+	+	+	+
Nematomorpha	+	+	+	+	+	+	+
Turbellaria	-	-	-	-	+	-	+
Gammaridae	-	+	+	+	+	+	-
Oligochaeta	+	-	+	+	+	-	+
Glossiphonidae	+	+	+	-	+	-	+
Erpobdellidae	-	-	-	-	-	-	+
Valvatidae	-	-	-	-	+	-	-
Planorbidae	-	-	-	-	+	-	-
Hydrobiidae	-	-	-	-	+	-	+
Ancylidae	-	-	-	-	+	-	+
Physidae	-	-	-	-	-	-	-
Corixidae	-	-	-	+	-	-	-
Ephemerellidae	+	+	+	+	-	-	-
Baetidae	+	+	+	+	+	+	-
Potamanthidae	+	-	-	+	-	-	-
Caenidae	+	-	-	+	-	-	-
Heptageniidae	+	-	-	+	-	-	-
Ephemeridae	+	-	-	+	-	-	-
Tipulidae	+	+	-	-	+	-	-
Dixidae	+	+	-	-	+	+	-
Dytiscidae	-	-	-	+	+	-	-
Gyrinidae	+	-	+	+	-	-	-
Trichoptera	+	+	+	+	-	-	+
Leuctridae	+	+	+	+	-	-	-
Perlodidae	+	+	+	+	-	-	-
Hemerobiidae	+	-	-	-	-	-	-
Haliplidae	+	+	+	+	+	+	+
Diptera	+	+	+	+	+	+	-
Simulidae	-	-	+	+	-	-	-
Mesoveliidae	-	-	-	+	-	-	-
Hydrosychidae	+	-	-	-	-	-	+
Asellidae	-	-	-	+	+	+	+
Notonectidae	**-**	**-**	**-**	**-**	**-**	**-**	**+**
Total Taxa	**20**	**13**	**14**	**21**	**17**	**8**	**13**

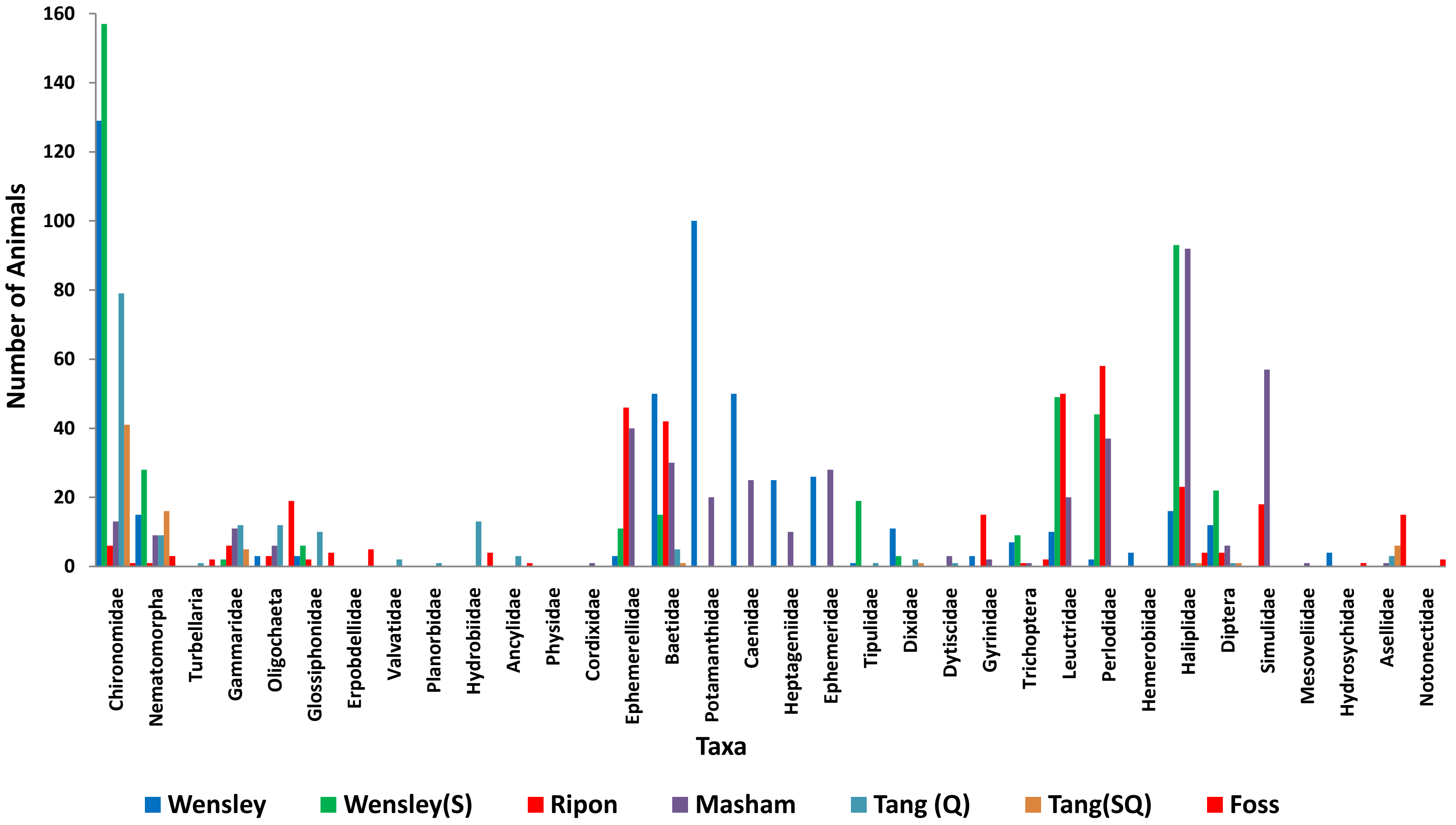


Figure 3.7. Community abundance of macroinvertebrates for different sites in the River Ure, Tang Beck and the River Foss. (S) represents summer data while (SQ) shows semi-quantitative samples

At Wensley, the number of taxa was lower in the summer. Some families of mayflies were absent which were present at Wensley in the spring. The number of taxa (total families) and the number of clean indicator species (individuals) recorded are higher in Wensley in the spring compared to the summer. Tolerant taxa make up 30% of the total population at Wensley in the summer but only 20% in the spring. The total number of individuals was less in summer (Figure 3.8). Differences were seen in the kind of fauna in quantitative and semi-quantitatively samples from Tang Beck. Many families were absent in semi-quantitative samples and there were fewer (<50%) individuals compared to samples collected quantitatively. The quantitative sampling approach yielded a high species (taxon) richness, higher numbers of indicator species and a higher community composition compared with samples collected using the semi-quantitative method (Table 3.2).

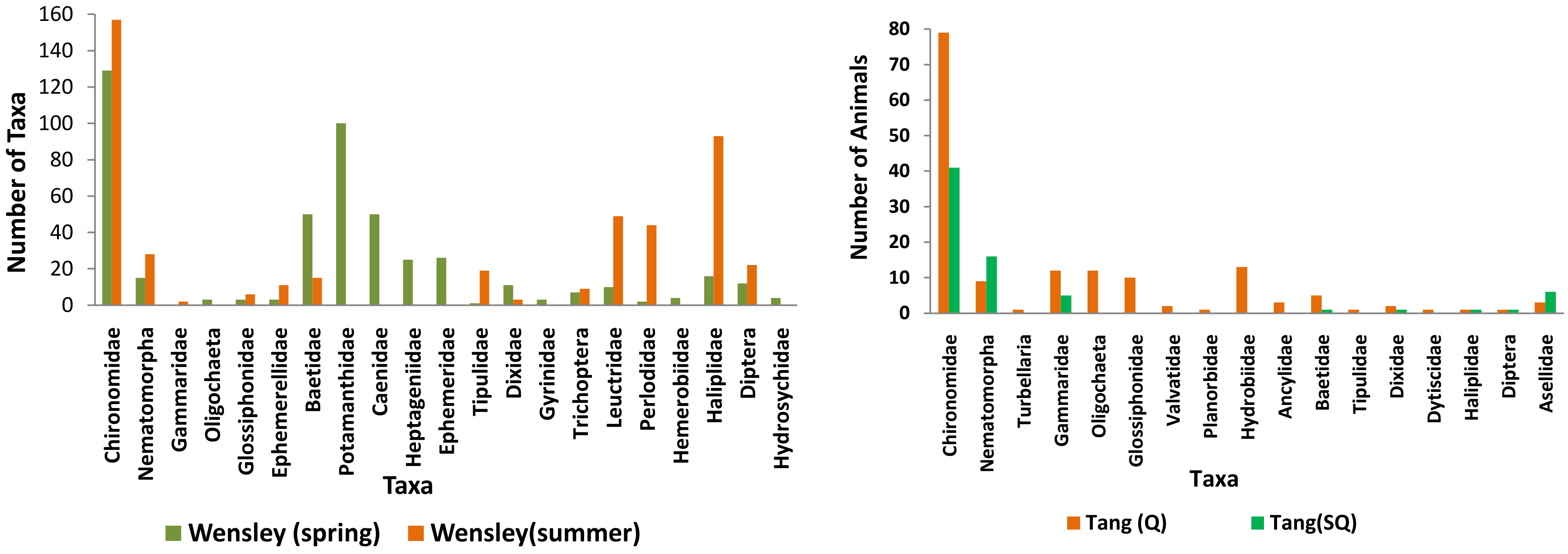


Figure 3.8. Left: Effect of season (spring and summer) on community composition at Wensley, Right: Effect of quantitative and semi-quantitative methods at Tang Beck.

The Biological Monitoring Working Party (BMWP) score (Table 3.4, Figure 3.9) indicates that the River Ure at Masham has the highest value followed by Wensley. The lowest value (43) was recorded in the River Foss for the spring samples. At Tang Beck scores were higher for quantitative compared to semi-quantitative samples. The BMWP score decreased at Wensley between spring and summer, from 124 to 77.

The Average Score per Taxon (ASPT) values for the sites (Figure 3.10) shows that Wensley has the highest value with a score 6.2. This site is not dominated by a single taxon, and the presence of higher numbers of very sensitive species and fewer numbers of pollution tolerant species is an indication of good quality habitat (Rosenberg *et al.*, 2004). ASPT is lower in Masham, Ripon and at Tang Beck, with the lowest score was recorded for the River Foss, indicating that the general water quality at the site is poor (Mason, 2002b).

The total ASPT value recorded at Wensley in the summer is lower than in the spring. However the lower score for summer is still higher than for Ripon and Tang Beck, which indicates that the quality of the River Ure in Wensley is comparatively good, even with seasonal changes taken into account. In contrast to BMWP, the ASPT score for the semi-quantitative method is higher than that for the quantitative method at Tang Beck (Fig. 3.10).

Table 3.4. BMWP scores for macro-invertebrates collected from the River Ure, Tang Beck and the River Foss. The tolerant species are given the least score while sensitive species are given a high score. (S) indicates summer, (Q) quantitative and (SQ) semi-quantitative data.

Taxa	Wensley	Wensley(S)	Ripon	Masham	Tang(Q)	Tang(SQ)	Foss
Chironomidae	2	2	2	2	2	2	2
Nematomorpha	1	1	1	1	1	1	1
Turbellaria	0	0	0	0	1	0	1
Gammaridae	0	6	6	6	6	6	0
Oligochaeta	1	0	1	1	1	0	1
Glossiphonidae	3	3	3	0	3	0	3
Erpobdellidae	0	0	0	0	0	0	3
Valvatidae	0	0	0	0	3	0	0
Planorbidae	0	0	0	0	3	0	0
Hydrobiidae	0	0	0	0	3	0	3
Ancylidae	0	0	0	0	6	0	6
Physidae	0	0	0	0	0	0	0
Corixidae	0	0	0	5	0	0	0
Ephemerellidae	10	10	10	10	0	0	0
Baetidae	10	10	10	10	10	10	0
Potamanthidae	10	0	0	10	0	0	0
Caenidae	7	0	0	7	0	0	0
Heptageniidae	10	0	0	10	0	0	0
Ephemeridae	10	0	0	10	0	0	0
Tipulidae	5	5	0	0	5	0	0
Dixidae	5	5	0	0	5	5	0
Dytiscidae	0	0	0	5	5	0	0
Gyrinidae	5	0	5	5	0	0	0
Trichoptera	5	5	5	5	0	0	5
Leuctridae	10	10	10	10	0	0	0
Perlodidae	10	10	10	10	0	0	0
Hemerobiidae	5	0	0	0	0	0	0
Haliplidae	5	5	5	5	5	5	5
Diptera	5	5	5	5	5	5	0
Simulidae	0	0	5	5	0	0	0
Mesoveliidae	0	0	0	5	0	0	0
Hydrosychidae	5	0	0	0	0	0	5
Asellidae	0	0	0	3	3	3	3
Notonectidae	0	0	0	0	0	0	5
Total Score	**124**	**77**	**78**	**130**	**67**	**37**	**43**

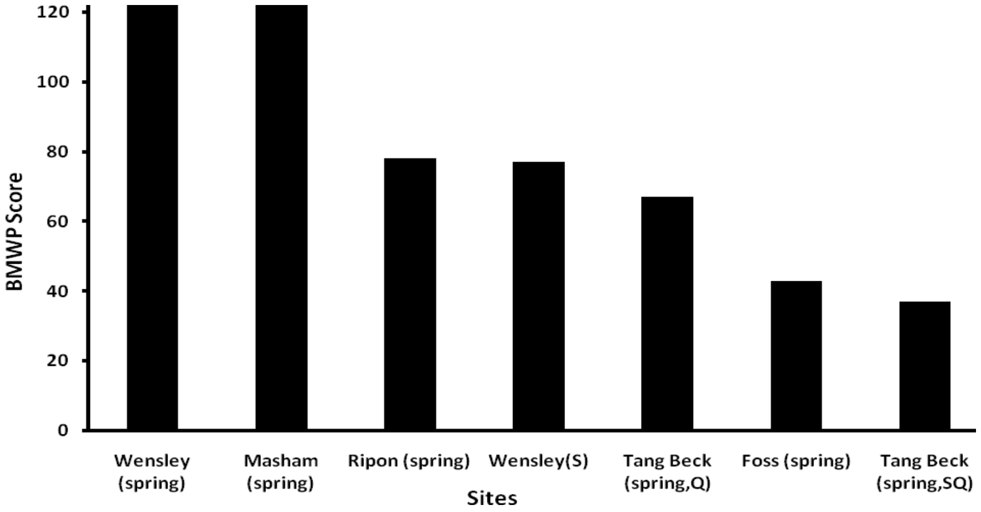


Figure 3.9. Biological Monitoring Working Party (BMWP) score for different sites on the River Ure Tang Beck and the River Foss. (S) is for summer and (SQ) is for semiquantitative data.

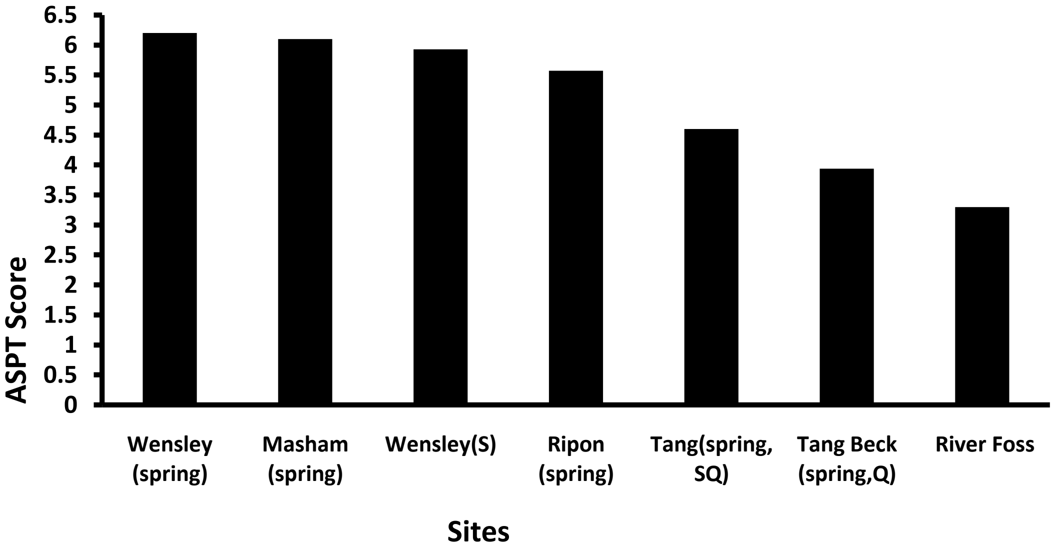


Figure 3.10. Average Score Per Taxon (ASPT) at different locations on the River Ure, Tang Beck and the River Foss, where (S) is for summer and (SQ) is for semi-quantitative samples.

3.5 Body Mass Patterns

Visual inspection of the body size histograms shows that all sites are skewed towards smaller body masses, except the Foss which shows clear multiple modes (Figure 3.11). The body size spectra for all sites are mainly dominated by individuals in the size categories less than 0.25- 1mg dry weight, but there is no obvious evidence for uni-modality in body mass distributions within any site. There are indications of other modes in the range 0.5-3mg and, at least in some sites, it seems likely that there is another mode at the largest size classes.

The data collected in spring often show a clear mode for large size groups and this was not seen for the summer data at Wensley (Figure 3.13), where a second mode appears at sizes of 1-2mg dry weight. The body sizes of animals collected at Tang Beck by semi-quantitative techniques appear uni-modal, or at least to have fewer modes when compared to those collected using quantitative methods. There is one distinct peak at a small body mass and one minor peak in the size class 1.25mg dry weight. Interestingly, the most polluted site, as determined from the ASPT, appeared to have multiple modes, but this could be due to there being fewer individuals at the site.

However visual inspection of histograms for detecting modality is unreliable because of possible bin size effects. Kernel density analysis of the spring samples revealed more than one significant mode for the cleaner sites and only a single mode for the Foss at Tang Beck (Figure 3.12, Table 3.5). However, in the summer sample for Wensley there was also only one mode. Comparison of size spectra derived from quantitative and semi-quantitative sampling showed differences in number of modes detected (quantitative = 1, semi-quantitative = 2).

In summary, for samples from sites sampled with a Surber sampler at the same time of year (spring), there is an indication that sites with better water quality have more modes in their body size spectra. However, there were also seasonal effects at Wensley, with fewer modes in summer, when the ASPT was also lower (Figure 3.13). The number of modes recorded was also sensitive to the sampling technique used, as shown for Tang Beck.

The relationship between water quality and modality for the spring samples is statistically significant for ASPT ($R^2 = 0.8552$, $p<0.05$ and for BMWP ($R^2 = 0.9216$, $p<0.05$) (Figure 3.14B), but not when summer and semi quantitative data are included ($R^2 =0.5635$, $p>0.05$) for BMWP and ($R^2 =0.4643$, $p>0.05$) for ASPT.

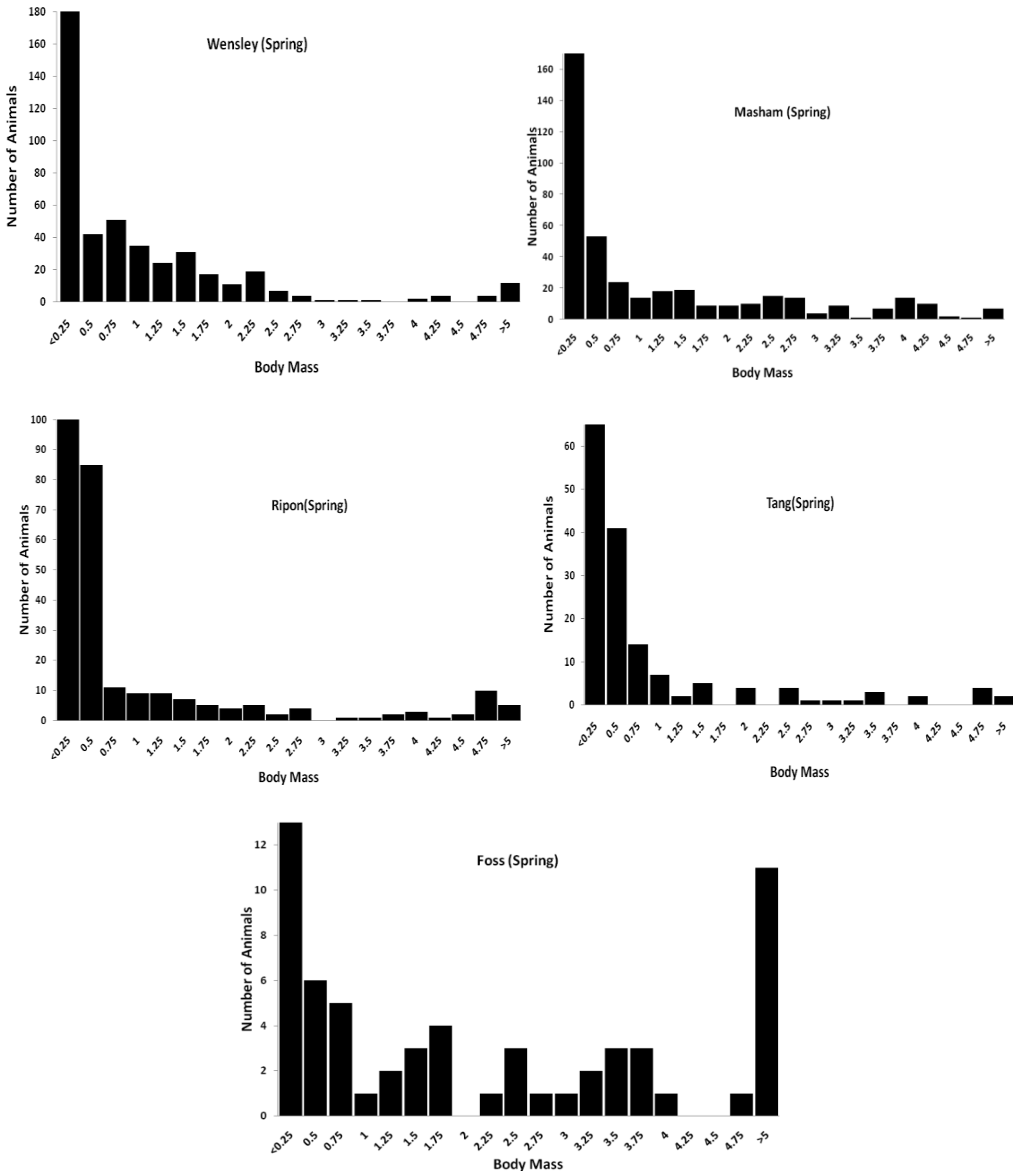


Figure 3.11. Frequency histograms of body mass distributions of streams communities for the River Ure, Tang Beck and the River Foss. Body mass of an animals were determined in mg.

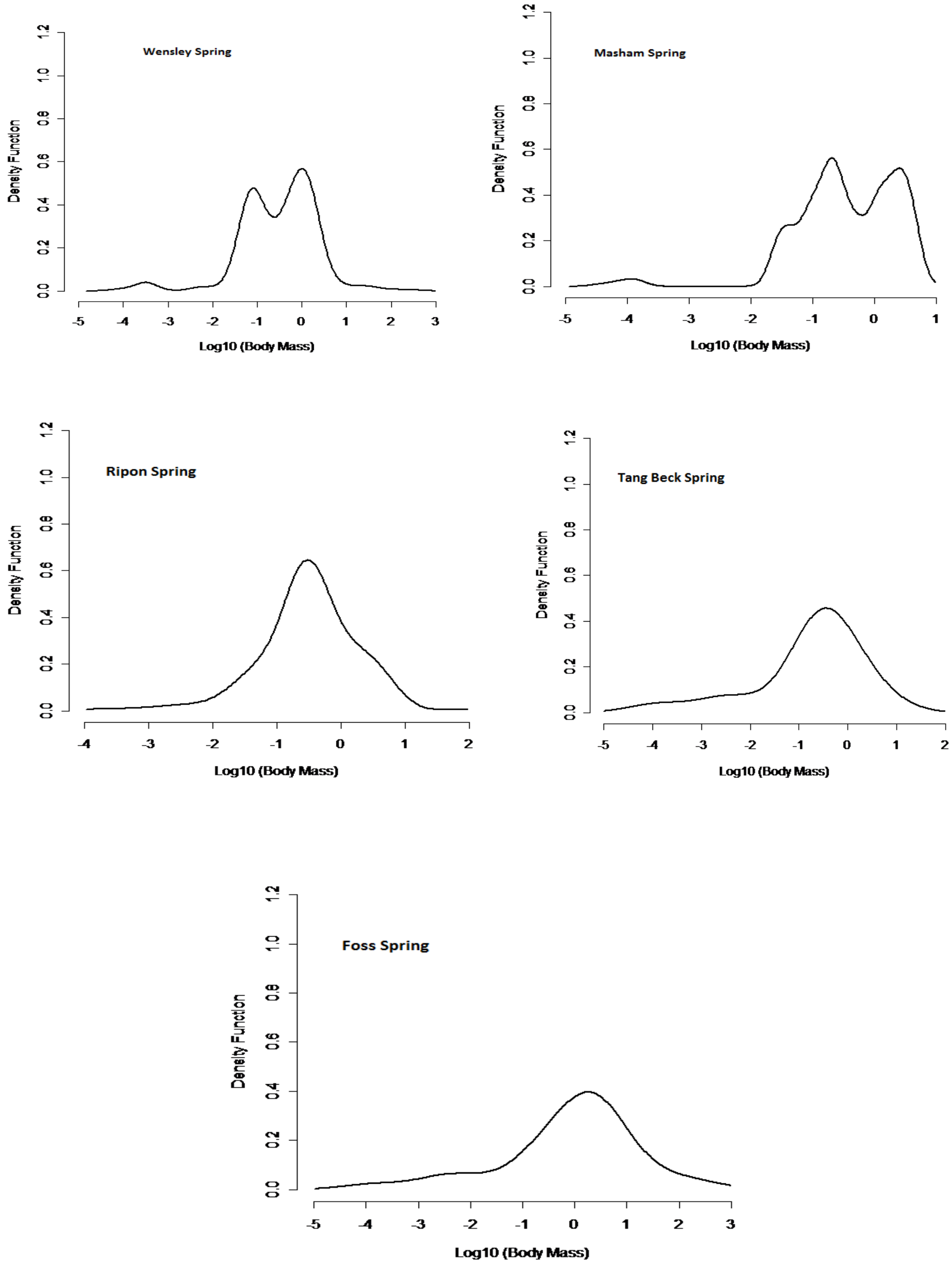


Figure 3.12. The fitted distributions for density-body size estimated by kernel density estimation and bootstrapped re-sampling for the River Ure, Tang Beck and the River Foss. Both axes are scaled as $\log_{10}$ of the original data. Density function = number of individuals each site.

Table 3.5. The results of the test of significance for kernel density estimation where h is a smoothing constant used in kernel density estimation, m is the smallest number of modes for which the bootstrap test was not significant at the 5% level, α is the level of significance for each distribution (number of modes).

Sites	**h**	**m**	***α***
Wensley (spring)	0.219	3	0.682
Wensley (summer)	0.325	1	0.089
Ripon (spring)	0.307	2	0.464
Masham (spring)	0.146	3	0.527
Foss (spring)	0.578	1	0.304
Tang (spring)	0.552	1	0.075
Tang (semi-quantitative)	0.238	2	0.452

Table 3.6. The locations of troughs in the kernel density analysis and the location of gaps using the BMDI techique.

Sites	Troughs (mg)		Gaps (mg)
Wensley	0.00151	↔	0.0013, 0.00359
	0.24	↔	0.247, 4.02
Masham	0.00181	↔	0.019
	0.63		
Ripon	-		0.002
			0.7
Tang Beck	-		0.0026
			0.04
Foss	-	↔	0.006
Wensley (summer)	-	↔	0.005
			0.013
			0.018
			0.021
			0.094
			0.57
			0.73
			1.03
			2.6
Tang Beck (semi- quantitative)	0.00102	↔	0.000939

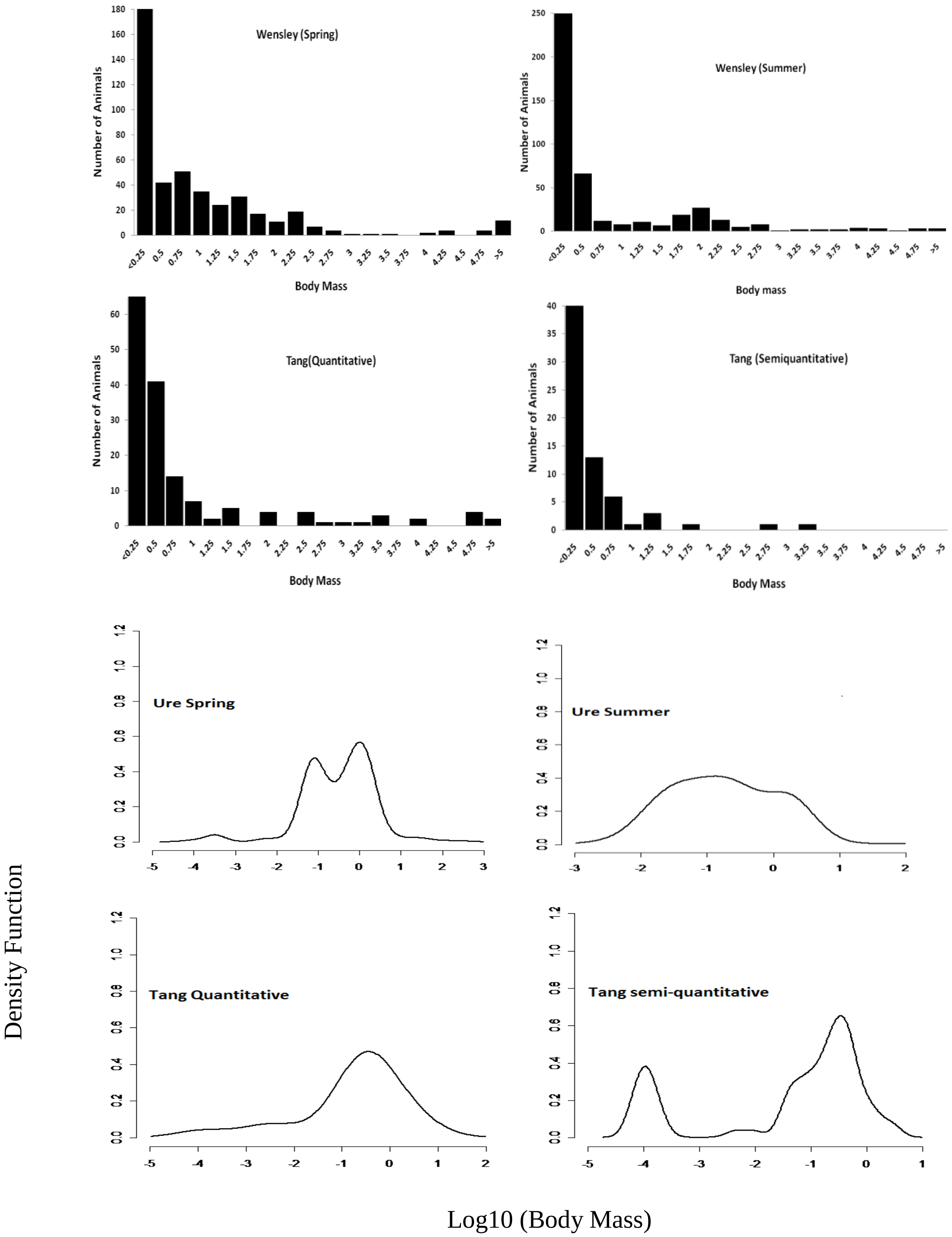


Figure 3.13. Frequency histograms and kernel density estimates for Wensley and Tang Beck, to evaluate body mass distributions in spring and summer and when using quantitative and semi quantitative methods. The number of modes found varied seasonally at Wensley and with sampling method at Tang Beck.

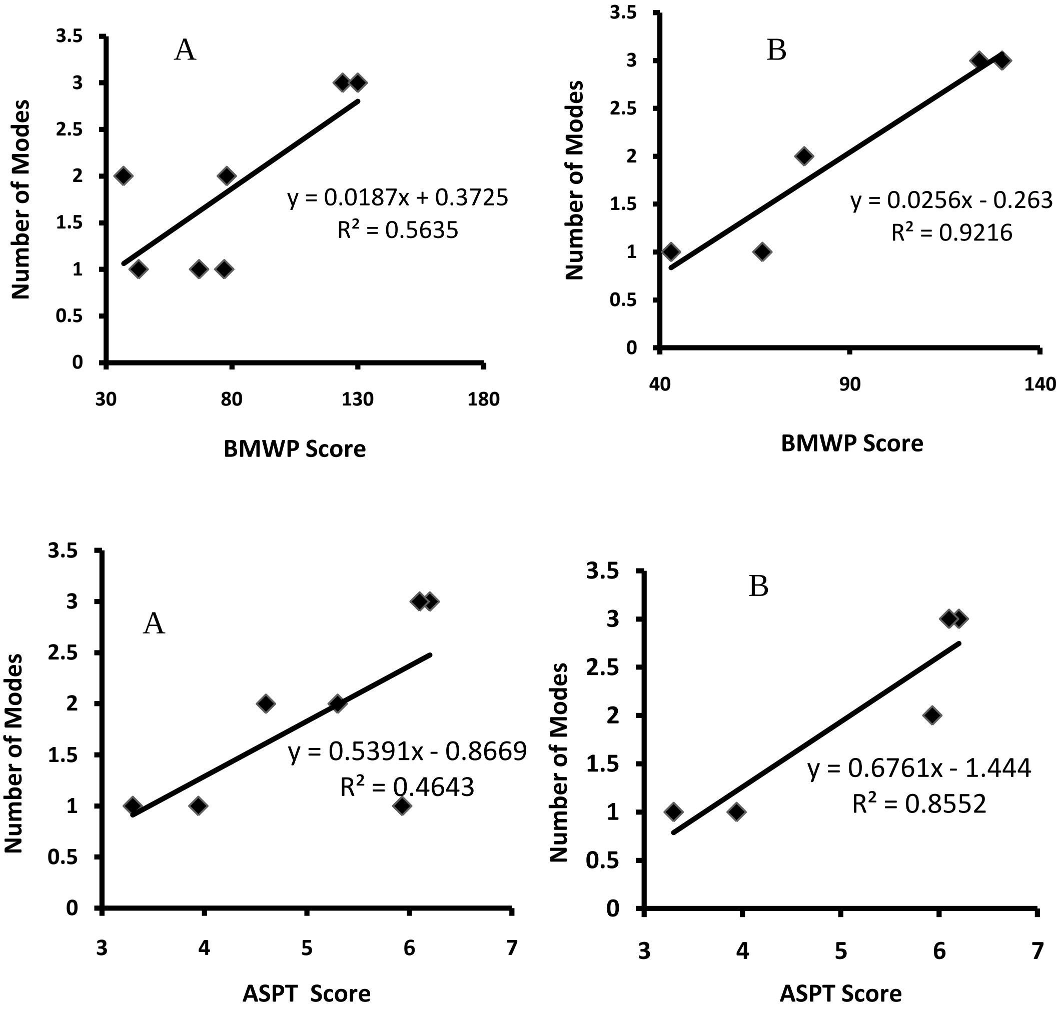


Figure 3.14. The relationships between water quality and biotic indices (BMWP, ASPT). A = including summer and semi-quantitative, B = spring and quantitative data only.

3.5.1 Body Mass Difference Index (BMDI)

Gap analysis for body mass distributions was carried out using Holling's approach, the BMDI (Holling, 1992) The highest number of gaps was detected for the cleanest site at Wensley and the lowest for the most polluted site on the River Foss (Figure 3.15). For intermediate quality habitats the analysis shows significant variation, such as at Masham which has the best water quality (ASPT and BMWP) after Wensley, but only one gap was detected in gap analysis. Two gaps were recorded for Ripon and Tang Beck, which have fairly good and fair quality water respectively. As a result, there, are no significant relationships between the number of gaps and water quality (Figure 3.17) neither for BMWP ($R^2 = 0.1667$, $P>0.05$) nor for ASPT ($R^2 = 0.2225$, $p>0.05$). Gap analysis seems to be disproportionately influenced by large values of BMDI, which increase the mean value used to determine the criterion line, so that gaps present within lower ranges could not be detected.

Contrary to the results for the number of modes, the number of gaps detected for Wensley in the summer was higher than in the spring. At Tang Beck, there was only one gap recorded in data collected using the semi- quantitative method but two gaps revealed in the data using a quantitative approach (Figure 3.16). The locations of gaps showed little relationship to the location of troughs determined by kernel density analysis (Table 3.6). The number of gaps at Wensley are 4, while only two troughs were detected by KDE. The locations of troughs at Wensley and Tang Beck (semi-quantitative) are similar with the locations of same gaps revealed by BMDI (Table 3.6).

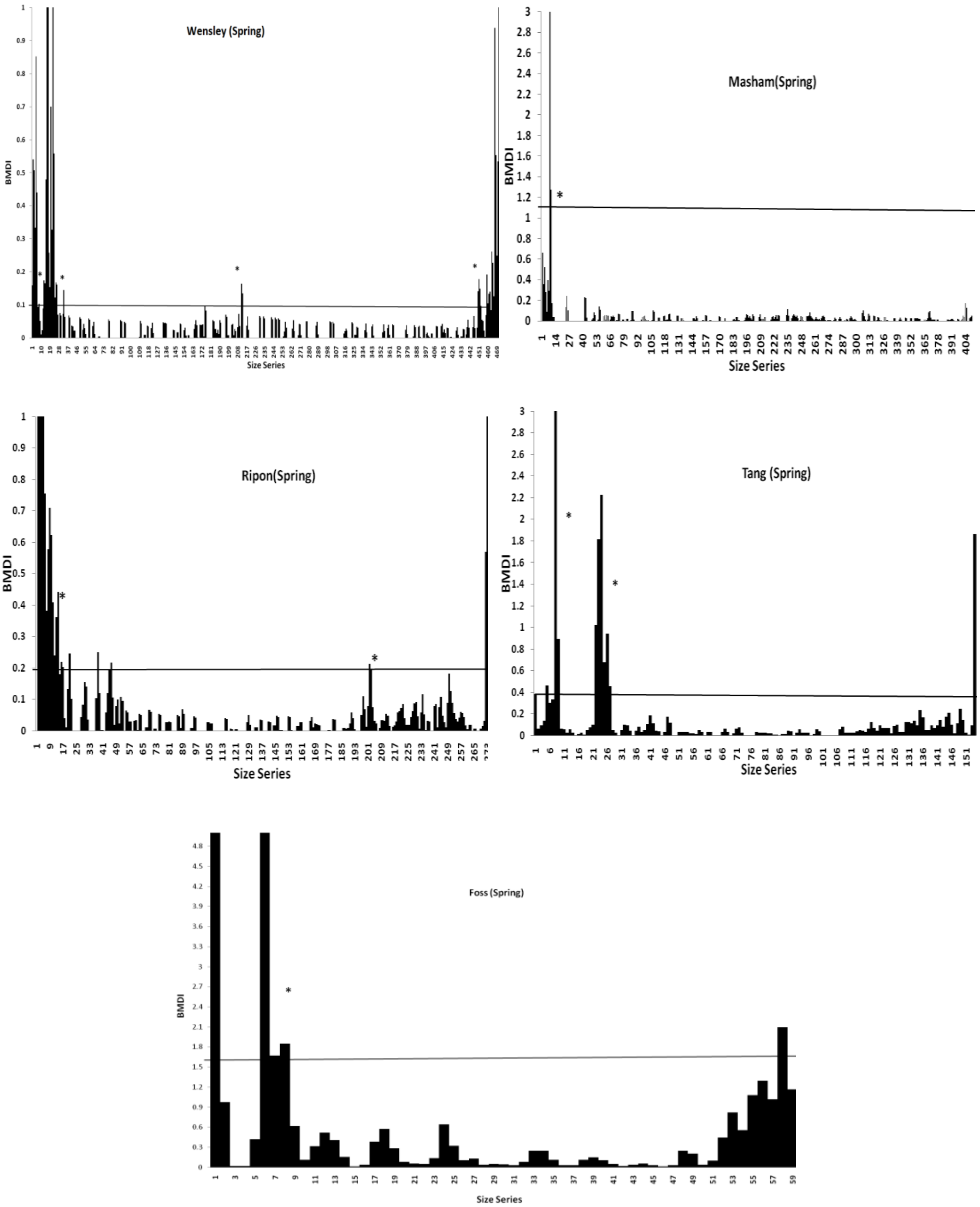


Figure 3.15. Distributions of body mass gaps for River Ure, Tang Beck and River Foss stream communities in spring. The horizontal line is the mean +2SE and asterisks (*) show significant body mass gaps, identified as at least two values of BMDI above and followed by four values below the line (Holling 1992).

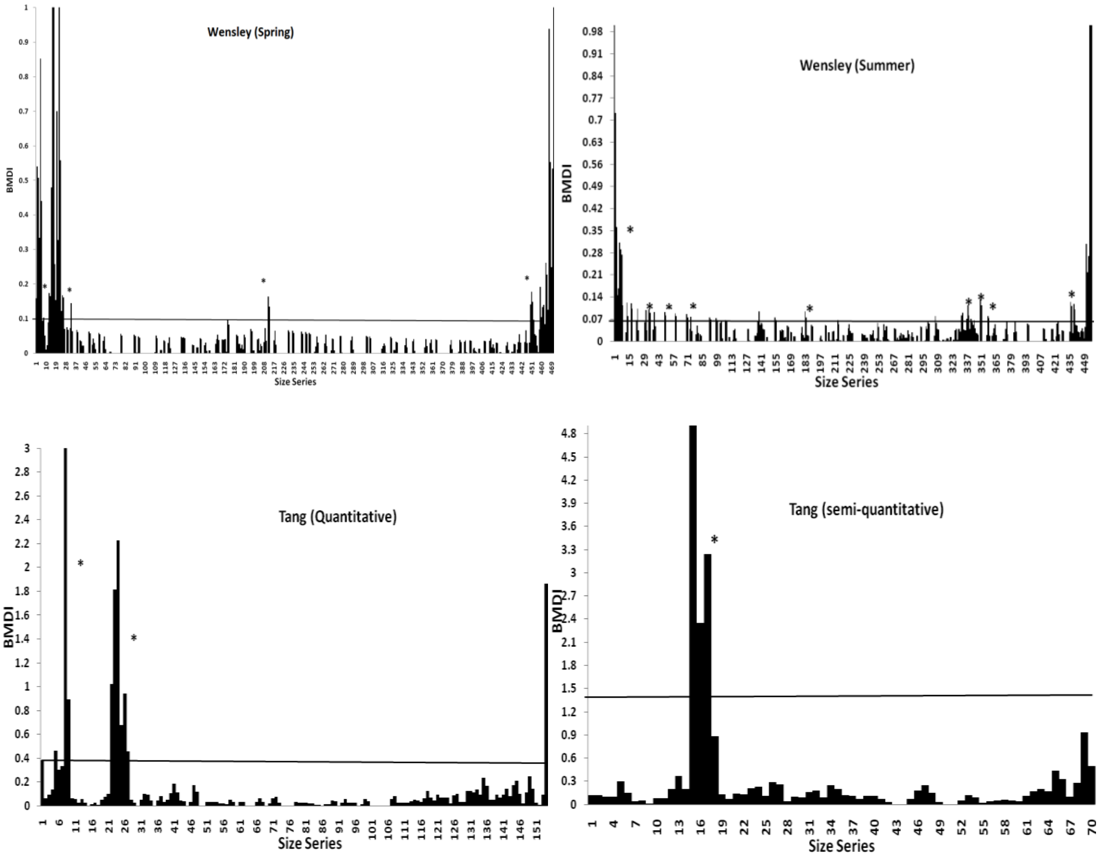


Figure 3.16 Upper: Body mass gaps in distributions at Wensley in spring and summer**.** Lower: Gaps for quantitative and semi-quantitative samples from Tang Beck. (*****) shows significant body mass gaps, identified as at least two values of BMDI above, followed by four values below the criterion line (mean+ 2SE) (Holling 1992).

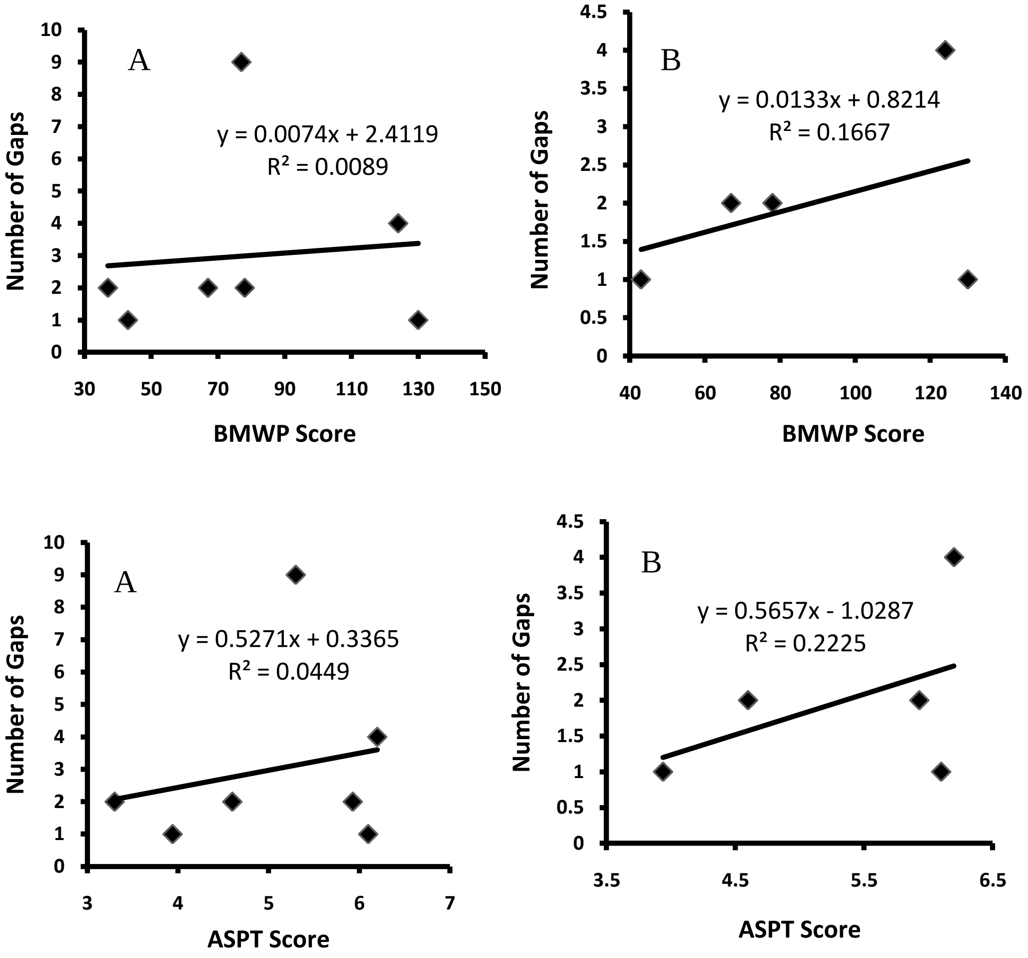


Figure 3.17 Relationships between the number of gaps and water quality as determined by biotic indices (BMWP, ASPT). A = summer and semi-quantitative, B for spring data only.

3.6 Discussion

3.6.1 Water quality and biotic indices

The use of invertebrates to determine the health of streams including their composition and community structure has been carried out for more than a century (Kolkwitz and Marsson, 1902), and in many parts of the world this method is well established (Hawkins, 2006; Buss and Salles, 2007). Three pollution index measures the Macroinvertebrates Community Index (MCI), Biological Monitoring Working Party Score (BMWPS) and Average Score per Taxon (ASPT) were calculated in the present study because biotic indices are a first approach which can be used to characterize the health of an ecosystem (Feio *et al.*, 2009).

3.6.2 Water quality across the sites

My analysis shows a clear gradient of water quality across the sites, with Wensley cleanest and Foss the most polluted, as determined by the abundance and composition of macroinvertebrates at the different sites. Thus, the highest numbers of individuals (474) was found at Wensley whilst fewer animals occurred on the Foss, with only 63 individuals present. The occurrence of benthic species in aquatic system is related to the kind of substratum and any adverse abiotic condition such as pollution may limit species richness (Singh and Gaur, 1989). Numerous studies have revealed that species richness and composition changes with environmental disturbance, mainly from pollution (Reash, 1990).

The composition of families varies among sites. The total number of taxa at Wensley was 20, and the fewest were recorded for the River Foss with only 13 families present (Table 3.2). Numbers of the indicator taxa EPT (Ephemeroptera, Plecoptera and Trichoptera) are high in Wensley while these taxa are fewer or absent in the River Foss. The species composition shows a higher proportion (56%) of EPT at Wensley with only 7% present on the River Foss. On the basis of species richness and composition, the condition of the River Ure at Wensley is appears to be of better or ‘good’ quality while the River Foss was categorized as ‘poor’. The validations of these categories are subjective and qualitative and may vary between geographical areas. However, BMWP analysis for the sites provides similar information with Wensley having a high score (124) and the least score being recorded for the River Foss, with a score 43 (Table 3.4, Figure 3.9). The BMWP scores were divided by the number of taxa to produce ASPT score, (Figure 3.10) revealing that Wensley has the highest value with a score of 6.2 while the River Foss has the lowest value of 3.3.

Masham, Ripon and Tang Beck have intermediate water quality and Masham and Ripon always ranked the same in term of their biotic indices. The number of individuals at Masham, Ripon and Tang Beck was 413, 275 and 156, respectively. Similarly, the number of families of macro-invertebrates at Masham was 21, followed by Ripon and Tang Beck with 14 and 17 taxa, respectively. However, the BMWP score for Masham was very high with 130, the highest score for the spring samples, while scores of 78 and 67 were recorded for Ripon and Tang Beck. The ASPT scores for sites with intermediate water quality were similarly ordered with a score of 6.1 for Masham, 5.93 for Ripon and 3.9 for Tang Beck.

On the basis of taxon composition, number of individuals and biotic index scores (Community Index, BMWP and ASPT), it can be concluded that Wensley has very good ecological status, Masham has good quality while Ripon is of fairly good quality. Tang Beck quality is fair and the River Foss site has poor quality water.

3.6.3 Effect of General Habitat Conditions

Vegetation structure and abiotic variables (water temperature, salinity, depth and dissolved oxygen) are important in regulating invertebrate presence, absence and species richness (Wirwa. and Nicholas, 2008). In the present study, the majority of stream habiats were stony riffles but differed in water depth. The River Ure at Wensley was shallow but at Masham and Ripon the water was slightly deeper (although still easily sampled by Surber). The biggest difference in substarte was at Tang beck where the stream bottom was quite muddy while river Foss passes through clay beds and is very turbid. Thus, the effect of substrate on the distribution and abundance of macro-invertebrates cannot be ruled out. Small scale variation in microhabitat complexity can result in different patterns of invertebrate distributions between fresh water habitats (Michael and Leland, 2005) and has been reported as affecting body size distributions in benthic communities (Robson *et al.*, 2005). Ideally, I could have explored theis further for these sites using RIVPACS data, as done in the next Chapter 4, but these data were not available for these sites. Thus, the following discussion needs to be interpeted with caution, especially when comparing the two muddier sites (Tang Beck and the Foss) with the three stony sites (Wensley, Masham and Ripon). However, habitat structure is not a factor when comparing the effects of season and sampling teachnique at the same site.

3.6.4 Body size distributions

Frequency histogram body mass distributions were right-skewed in all sites. None of the sites clearly showed unimodality and there was always a clear mode for larger size classes in all sites. The River Foss histograms indicate multimodality, but this may reflect the fewer individual recorded at this site. Changes in modality with water quality were not obvious from the frequency histogram and more rigorous analysis using BMDI and KDE were therefore carried out.

3.6.5 BMDI approach

The number of gaps detected using the BMDI approach in high and poor quality sites was similar for both BMWP and ASPT measures of water quality (Figure 3.15). At Wensley, the numbers of gaps was 4, while only 1 gap was detected for the River Foss, which has poor quality water. For intermediate quality sites, the number of gaps is more variable, with only one gap for Masham, which has good quality and in Ripon and Tang Beck the two gaps were recorded which have fairly good and fair ecological status. For intermediate water quality sites it seems unlikely that differences in sediment architecture at this scale is responsible for this variation. In the BMDI analysis large consecutive differences in body mass produces high values of the index, ultimately increasing the mean index value used to produce the criterion line so that it become impossible to detect gaps which may occur among lower BMDI values.

3.6.6 KDE approach

Silverman's bump hunting technique (Silverman, 1981) to determine modality is considered a conservative test. This method estimates the number of modes required to determine each distribution (Holling and Allen, 2002). The very clean and intermediate quality sites indicated at least bimodality (Figure 3.12, Table 3.5). In contrast to the right skewed size distributions in other mass spectra (Blackburn and Gaston, 1994) our findings, for clean and intermediate sites (Wensley and Masham) revealed three modes in their body size distributions. Tang Beck and the River Foss had only a single mode. The KDE approach revealed the evidence for changes in modality with water quality, but the locations of troughs in this analysis were different that the locations of gaps recorded by BMDI analysis (Table 3.6). At Wensley the locations of troughs are at 0.00151 and 0.24 mg and gaps at 0.0013 and 0.24mg are similar, but BMDI defined two extra gaps at this site. Similarly, for Masham two troughs were indicated by KDE at 0.00181 and 0.63 mg, but only one gap was detected at 0.019 mg by BMDI. Ripon and Tang Beck which are unimodal and have no trough, while River Foss has a trough at 0.00841 and gap at 0.006 mg. KDE indicates no troughs for Wensley in the summer but the same data set had 9 gaps when analysed by BMDI. Tang Beck (semi- quantitative) had a trough and gap locations at 0.001 and 0.000939 mg respectively. Thus, overall my results may provide no consistent support for a relationship between these two techniques. Usually the troughs defined by KDE had an equivalent gap, but not all the gaps detected by BMDI were revealed as troughs by KDE.

3.7 Relationships between number of gaps, number of modes and water quality

There was no clear relationship between the number of gaps and water quality (Figure 3.17B) for both BMWP and ASPT analysis, but there was a relationship between water quality and number of modes and this was highly significant, implying that body mass distributions may reflect environmental disturbance. However, the effects of habitat structure cannot be ruled out for this data set and this finding should be interpreted with caution. Chapter 4 explores further the effects of potentially confounding habitat variables within the same river system (Aire) where samples were taken at the same time of year. The distribution of body mass in an ecosystem could have potential as a bioassay of different environmental and ecological processes. However, the effects of habitat structure cannot be ruled out for this data set since there is a confounding variable of substratum type: the most disturbed site in terms of pollution also has more of clay and less of a stony substratum. This is often an issue with freshwater systems, where sedimentation can be much higher in more polluted sites. In the present study it is not possible to exclude this site from the analyses with also compromising statistical power. However, the kind of organisms present at the Tang Beck site are similar to those which might be expected from disturbed streams with stony substrata (they are not soft sediment biota). This finding should therefore be interpreted with caution at this stage and viewed against the results from the next chapter, chapter 4, where as many of such confounding variables as possible are removed from the analyses by selecting a single river system and sampling at the same time of year and at sites which were all stony.

There are many alternative explanations of modality in body mass distributions, such as community interactions (Hutchinson, 1959); energetic hypothesis, based on energy availability (Lovegrove and Haines, 2004); evolutionary histories (Smith *et al.*, 2004); biogeographical histories (Silva *et al.*, 2001). An alternative hypothesis to all of these is the Textural Discontinuity Hypothesis (TDH) based on the observation that discontinuous distributions of body mass are due to ecological processes that occur in systems discontinuously. The occurrence of these discontinuous processes is regarded as a sign of a resilient ecosystem (Holling, 1992) and several studies have found such patterns of body mass in a variety of habitats (Allen *et al.*, 1999; Fu *et al.*, 2004).

In this chapter, multiple modes and gaps in the cleanest site and fewer modes and gaps in the polluted site, indicate different ecological processes acting at each site with respect to the above hypotheses. There is no geographical boundaries that restrain animal movement in the river and the invertebrate communities, present are broadly similar across the sites so that a phylogenetic explanation is unlikely, as is the community interaction hypothesis. The remaining hypothesis, the Textural Discontinuity Hypothesis is the only one left which could explain better the presence of multiple gaps and modes at the cleaner site and one mode or gap in the most polluted habitat (River Foss). However, these clean and dirty sites were from different river systems, so that biogeographical explanation cannot be ruled at altogether. For this reason a more detailed analysis was carried out within a single river system, the River Aire, all samples being taken with the same technique (Surber) in the same season (early autumn) as described in chapter 4.

3.8 Seasonal and methodological variations

Significant changes in taxon richness, diversity and abundance with respect to season have been found at Wensley for the spring and summer, with lower diversity and richness in the summer compared to the spring (Table 3.2). These differences are reflected in the BMWP and ASPT scores which were lower in summer (Table 3.4). The right-skewed frequency mass distributions among small and intermediate size classes (<0.02-2mg) for summer data indicates the dominance of small size individuals. This is consistent with the negative relationship between ASPT values and the temperature as revealed by Zamora et al (1995) or it may be that in late summer the water flows are generally low and polluted effluent become more concentrated reducing populations of susceptible species and hence also the ASPT score. Significant differences occurred in species richness, diversity and modality of body mass distributions at Tang Beck for semi-quantitative data (Table 3.2 and Table 3.3). BMDI revealed only one gap for the site using the semi-quantitative method, which indicates weakness of this approach in distinguishing stream qualities, as has been found in previous studies (Lenat, 1988).

In conclusion, size distributions of organisms have potential as metrics of community and ecosystem structure (Hanson *et al.*, 1989a). Statistically rigorous analysis of the number of modes (by KDE) indicates unimodality for the summer data (Figure 3.13, Table 3.5) while the data appear trimodal in the spring, in agreement with Schmid et al (2002) who claimed that the shape of species shift from bimodal to unimodality.

3.9 References

Allen, C. R., Forys, E. A. and Holling, C. S. (1999). Body mass patterns predict invasions and extinctions in transforming landscapes. *Ecosystems* 2(2): 114-121.

Anonymous (1997). In *Local environmental agency plan Swale, Ure and Ouse: Consultation report*, 144.

Anonymous (2009). York's rivers. City of York Council.

Armitage, P. D., Gunn, R. J. M., Furse, M. T., Wright, J. F. and Moss, D. (1987). The use of prediction to assess macroinvertebrate response to river regulation. *Hydrobiologia* 144(1): 25-32.

Bakker, V. J. and Kelt, D. A. (2000). Scale-dependant patterns in body size distributions of neotropical mammals. *Ecology* 81(12): 3530-3547.

Blackburn, T. M. and Gaston, K. J. (1994). Animal body size distributions: Patterns, mechanisms and implications. *Trends in Ecology & Evolution* 9(12): 471-474.

Buss, D. and Salles, F. (2007). Using baetidae species as biological indicators of environmental degradation in a Brazilian river basin. *Environmental Monitoring and Assessment* 130(1): 365-372.

Cooper, A. H. and Waltham, A. C. (1999). Subsidence caused by gypsum dissolution at Ripon, North Yorkshire. *Quarterly Journal of Engineering Geology and Hydrogeology* 32(4): 305-310.

Defra, U. (2010). Sensitive areas (eutrophic):

Http://webcache.Googleusercontent.Com/search?Q=cache:Lo8hyuhl1ej:Www.Defra.Gov.Uk/environment/quality/water/waterquality/sewage/sensarea/eutrophicnitrates.Htm+river+foss+is+eutrophic+sensitive&cd=1&hl=en&ct=clnk&gl=uk.

Feio, M. J., Almeida, S. F. P., Craveiro, S. C. and Calado, A. J. (2009). A comparison between biotic indices and predictive models in stream water quality assessment based on benthic diatom communities. *Ecological Indicators* 9(3): 497-507.

Fu, C., Wu, J., Wang, X., Lei, G. and Chen, J. (2004). Patterns of diversity, altitudinal range and body size among freshwater fishes in the Yangtze River basin, China. *Global Ecology & Biogeography* 13(6): 543-552.

Hall, P. and York, M. (2001). On the calibraion of Silvermans's test for multimodality. *Statistica Sinica* 11: 515-536.

Hanson, J. M., Prepas, E.E. and Mackay, W.C. (1989). Size distribution of macro-invertebrates communities in fresh water lake. *Canadian Journal of fishries and aquatic science* 46: 1510-1519.

Havlicek, T. D. and Carpenter, S. R. (2001). Pelagic species size distributions in lakes: Are they discontinuous? *Limnology and Oceanography* 46(5): 1021-1033.

Hawkes, H. A. (1998). Origin and development of the biological monitoring working party score system. *Water Research* 32(3): 964-968.

Hawkins, C. P. (2006). Maintaining and restoring the ecological integrity of freshwater ecosystems: Refining biological assessments. *Journal of Ecological Applications* 16: 1249.

Holling, C. S. (1992). Cross-scale morphology, geometry, and dynamics of ecosystems. *Ecological Monographs* 62(4): 447-502.

Holling, C. S. and Allen, C. R. (2002). Adaptive inference for distinguishing credible from incredible patterns in nature. *Ecosystems* 5(4): 319-328.

Hutchinson, G. E. (1959). Homage to Santa Rosalia, or why are there so many kinds of animals? *The American Naturalist*, 93: 245-249.

ISO (1979). Assessment of the biological quality of rivers by a macroinvertebrates 'score'. *International Standards Organization ISO/TC 147/SC 5/WG 6 N 5*: 18.

Kamenir, Y., Dubinsky, Z. and Zohary, T. (2004). Phytoplankton size structure stability in a meso-eutrophic subtropical lake. *Hydrobiologia* 520(1): 89-104.

Karr, J. R. (1996). *Rivers as sentinels: Using the biology of rivers to guide landscape managment. Pp in Naiman and Bilby, eds. The ecology and managment of streams and rivers in the pacific Northwest coastal ecoregion.* Springer - Verlag, New York.

Kolkwitz, R. and Marsson, M. (1902). Grundsätze für die biologische beurtheilung des wassers nach seiner flora und fauna, *Mittheilungen aus der königlichen prüfungsanstalt für wasserversorgung und abwässerbeseitung zu Berlin*. (1): 33-72.

Lenat, D. R. (1988). Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. *Journal of the North American Benthological Society* 7(3): 222-233.

Lewis, D. R., Williams, R. J. and Whitehead, P. G. (1997). Quality simulation along rivers (quasar): An application to the Yorkshire Ouse. *Science of The Total Environment* 194-195: 399-418.

Lovegrove, B. G. and Haines, L. (2004). The evolution of placental mammal body sizes: Evolutionary history, form and function. *Oecologia* 138(1): 13-27.

Mason, C. F. (2002). *Biology of fresh water pollution.*: Pearson Education Limited.

Metcalfe-Smith, J. L. (Ed) (1996). *Biological water quality assessment of running water based on macro-invertebrates communities.* In Petts, G and Calow, P.(eds) Rive*r Restoration,* . Blackwell, Oxford.

Metcalfe, J. L. (1989). Biological water quality assessment of running waters based on macroinvertebrate communities: History and present status in Europe. *Environmental Pollution* 60(1-2): 101-139.

Pinder, L. C. V., Ladle, M., Gledhill, T., Bass, J. A. B. and Matthews, A. M. (1987). Biological surveillance of water quality. A comparison of macroinvertebrates surveillance method in relation to assessment of water quality, in a Chalk streams. *Arch. Hydrobiologie* 109: 207-226.

Raffaelli, D., Hall, S., Emes, C. and Manly, B. (2000). Constraints on body size distributions: An experimental approach using a small-scale system. *Oecologia* 122(3): 389-398.

Reash, R. (1990). Physicochemical factors affecting the abundance and species richness of fishes in the Cimarron River. *Proceedings of the Oklahoma Academy of Science* 70: 23-28.

Rosenberg, R., Blomqvist, M., Nilsson, H., Cederwall, H. and Dimming, A. (2004). Marine quality assessment by use of benthic species-abundance distributions: A proposed new protocol within the European Union Water Framework Directive. *Marine Pollution Bulletin* 49(9-10): 728-739.

Rutt, G. P., Pickering, T. D. and Reynolds, N. R. M. (1993). The impact of livestock farming on Welsh streams: The development and testing of a rapid biological method for use in the assessment and control of oganic pollution from farms. *Environmental Pollution* 81: 217-228.

Schmid, P. E., Tokeshi, M. and Schmid-Araya, J. M. (2002). Scaling in stream communities. *Proceedings. Biological sciences The Royal Society* 269 (1509): 2587-2594.

Silva, M., Brimacombe, M. and Downing, J. A. (2001). Effects of body mass, climate, geography, and census area on population density of terrestrial mammals. *Global Ecology and Biogeography* 10(5): 469-485.

Silverman, B. W. (1981). Using kernel density estimates to investigate multimodality. *Journal of the Royal Statistical Society. Series B (Methodological)* 43(1): 97-99.

Singh, A. and Gaur, J. (1989). Algal epilithon and water quality of a stream receiving oil refinery effluent. *Hydrobiologia* 184(3): 193-199-199.

Smith, F. A., Brown, J. H., Haskell, J. P., Lyons, S. K., Alroy, J., Charnov, E. L., Dayan, T., Enquist, B. J., Ernest, S. K. M., Hadly, E. A., Jones, K. E., Kaufman, D. M., Marquet, P. A., Maurer, B. A., Niklas, K. J., Porter, W. P., Tiffney, B. and Willig, M. R. (2004). Similarity of mammalian body size across the taxonomic hierarchy and across space and time. *The American Naturalist* 163(5): 672-691.

Zamora-Muñoz, C., Sáinz-Cantero, C. E., Sánchez-Ortega, A. and Alba-Tercedor, J. (1995). Are biological indices BMPW and ASPT and their significance regarding water quality seasonally dependent? Factors explaining their variations. *Water Research* 29(1): 285-290.

Chapter 4

Body size patterns in the River Aire: a test of Holling's Textural Discontinuity Hypothesis

4.1 Abstract

This paper explores the relationship between water quality and body size patterns in stream communities in order to establish the potential of size based indicators for assessing environmental conditions. In addition, Holling's (1992) proposition that lumpiness occurs in body size distributions across a broad range of spatial and temporal scales is explored. Samples of stream benthos were collected at different stations in the River Aire, Yorkshire, UK, which vary in water quality. All sites showed skewed distributions towards smaller size classes and visual inspection indicated that most had two very obvious modes at medium and large size classes, except for most polluted habitats. Analysis of the number of gaps, using Holling's (1992) BMDI, revealed wide variation in clean and intermediate water quality sites, though the most polluted site had the fewest gaps. However, other disturbed sites had more gaps and some clean sites had fewer gaps. It is clear that size distributions in stream communities are lumpy, in the sense that most sites showed more than one mode or many gaps, but the number of gaps (discontinuities) is not correlated with disturbance, at least for freshwater quality. The Kernel Density Estimation (KDE) results revealed more than one significant mode for the cleaner sites and a single mode for the most polluted site, but overall the relationship between water quality and number of modes remained non-significant.

4.2 Introduction

Body size is an important parameter in ecological studies (Blackburn and Gaston, 1994), being a key determinant of a wide range of ecological and evolutionary parameters. Body size determines many aspects of life history such as metabolic efficiency, generation time and metabolism (Morse *et al.*, 1988). The relationship between body size and abundance has been shown to be a useful tool for describing patterns across a wide range of taxa and habitats (Blackburn and Gaston, 1999) with a diversity of biotic and abiotic factors influencing these patterns (Maurer and Brown, 1988; Cyr *et al.*, 1997). Departures from expected body mass distributions may provide an indication of disturbance in communities and insights into resilience (Damuth, 1992).

One approach to exploring body size and resilience has been developed by (Holling, 1992). His Textural Discontinuity Hypothesis (TDH) proposes that organisms develop specific physical and behavioural characteristics in response to environmental texture which varies across scales and which is reflected in discontinuities in organism body size distributions. In a wide range of marine ecosystems, the body size distribution of benthic organisms appear tri-modal (Schwinghamer, 1988) and in planktonic systems biomass size spectra models indicate that size distributions are also multi-modal (Sheldon *et al.*, 1972; Thiebaux and Dickie, 1992). Such discontinuities in the distributions of body size indicates self-organizing processes within ecosystems and may provide a tool to assess ecosystem resilience (Allen *et al.*, 2005).

In addition to Holling's TDH, there are multiple competing hypotheses regarding the determinants of body mass distributions of species. These include community interactions (Hutchinson, 1959) and related ecological processes (Brown *et al.*, 1993); the energetic hypothesis based on the allocation of energy for species growth and reproduction processes which are limited by the energy availability from the environment and by the subsequent transformation of energy into offspring (Allen *et al.*, 2006; Marquet *et al.*, 1995; Lovegrove and Haines, 2004); the phylogenetic hypothesis, reflecting different evolutionary histories of species (Smith *et al.*, 2004;

Cassey and Blackburn, 2004); and the biogeographical hypothesis, which suggests that multiple modes in body size distributions are due to the restricted set of species present in a given community (Silva *et al.*, 2001). Many studies have found a relationship between body mass distributions and geographical range (Gaston and Blackburn, 1996; Pyron, 1999).

The Textural Discontinuity Hypothesis (TDH) argues that species respond to biotic and abiotic processes across different scales (micro, meso, and macro–scales) in time and space and that this produces discontinuous distributions in their body sizes (Holling, 1992). Many authors have argued that different landscapes and biomes with different ecological structures produce different patterns of body size distributions (Allen *et al.*, 1999; Havlicek and Carpenter, 2001) and studies on freshwater fish found a relationship between gaps in body size distributions and habitat structure (Fu *et al.*, 2004). If the body mass pattern is controlled by landscape architecture, differences in phylogenetics, biogeography, energetics, and community interactions should not significantly change patterns in body mass configuration (Allen *et al.*, 2006).

Here, I explore these hypotheses in relation to disturbance in freshwater stream communities. Specifically I will:

1. Establish the modality of benthic body size distributions across a gradient of environmental stress (sites with differing water quality) within a single river system, using a standardised methodology and at the same time of year. Chapter 3 showed that relationships may be obscured by including samples from different seasons and the cleanest and dirtiest sites were from two different river systems, so that biogeography issues cannot be discounted. In addition, the most disturbed sites had, as is often the case, a more muddy substratum, whilst the least disturbed sites had stony riffle beds, further complicating between site comparisons. The sites chosen here were of a similar habitat type and RIVPACS data were available for all of them.

2. Explore the relationship between the number of gaps in body size distributions and water quality using a gap finding approach proposed by Holling

(1992) as well as Silverman's Kernel Density Estimation technique for finding modes.

3. Assess whether the modality and number of gaps has potential for assessing the ecological health (resilience) of streams.

4.3 Study sites

Nine locations in the River Aire catchment were chosen on the basis of their water quality, previously classified by the Environmental Agency. The range covers sites from very good to bad quality. These GQA grades (General Quality Assessment) had been previously established by the Environmental Agency for each site. The sites with their grading are given in Table No 1, while short descriptions of each site are provided below.

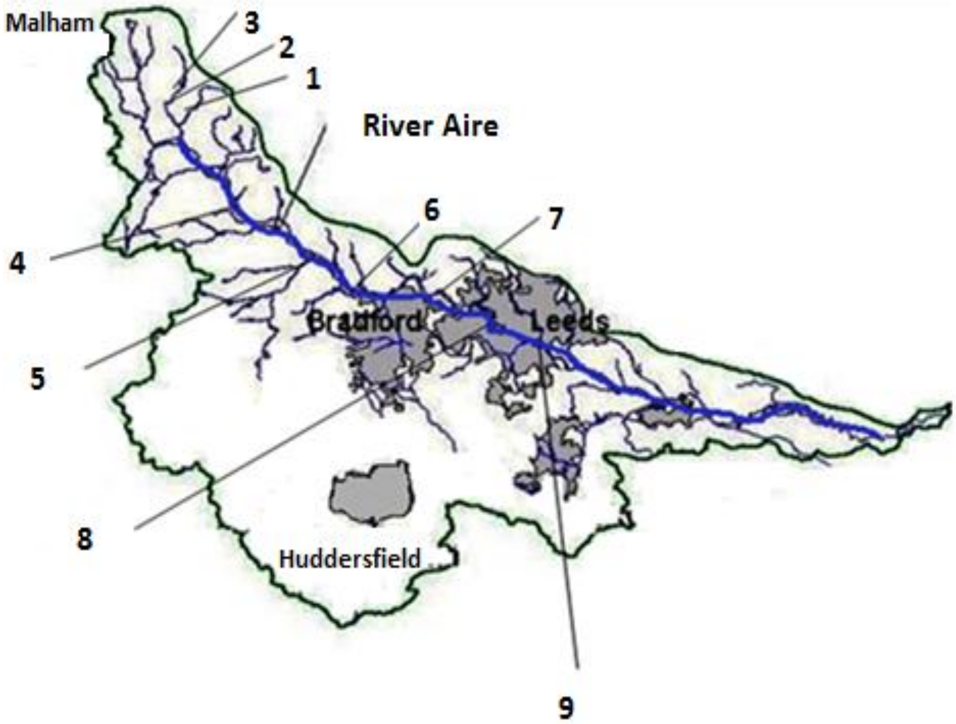


Figure 4.1. The dark blue line indicates the River Aire, flowing from North to South and the numbers 1-9 are sampling sites from Winterburn (very clean) to Thwaites mill (very poor).

Table 4.1.Sites sampled on the River Aire and their general grade as assessed by the Environment Agency.

No	Sites	General water quality
1	Winterburn	Very good
2	Airton	Very good
3	Otterburn	Good
4	Hetton	Good
5	Gargrave	Fair
6	Carlton Bridge	Fair
7	Esholt Village	Fairly good
8	Calverley Bridge	Poor
9	Thwaites Mill	Bad

1. Winterburn

Winterburn is a joint township with Flasby in the parish of Gargrave in North Yorkshire and is located 6 km south of Grassington. It is famous for its water reservoir which is one of several built in the Victorian era. Low lying pretty pasture lands characterize the bank up to Airton. The area is dominated by the influence of limestone and the combination of wildlife habitats and cultural features make the area particularly special. The River Aire has its origins above Malham cove and flow southerly direction for 6-7km and passes through the township Flasby Hall which has very good quality water.

2. Airton

Airton is a small village in the Craven district of North Yorkshire, situated 16 km North West of Skipton. Stone walls on the hillside show that farming has gone on here for hundreds of years. The River Aire passes through Airton before it reaches the town of Skipton. The water quality of the River Aire at this spot is very good.

3. Otterburn

Otterburn is a small village in the Craven district of North Yorkshire located 3km away from Airton. The area is predominantly upland moorland which supports a wide range of habitat types and wildlife. Clean water and a good supply of fish provide good habitat for otters.

4. Hetton

Hetton is a small village situated in North Yorkshire. Samples were collected from Flasby beck, which is also known as Hetton beck. The area is mostly grassland. The site is a clean and well maintained landscape of improved pasture which is divided by hedges. The water quality of this beck is considered very clean.

5. Gargrave

The area is located on the side of the Yorkshire Dales. The River Aire flows through the centre of village where it is surrounded by trees, presents beautiful scenery. Other water ways, the Leeds Liverpool canal, also pass through the village. The area is popular for hiking. Water quality is assessed as good.

6. Carlton Bridge

Carlton is a village in Richmondshire located in the Yorkshire Dale National Park near the River Cover. The place is quiet and provides excellent spots for walking and is accessible from the A444. The River Aire here is of fair quality.

7. Esholt Village

Esholt is a small village in west Yorkshire. The river passes through the industrial towns of Binglay and Shipley and meets Bradford Beck and heads past Esholt village. Due to domestic and industrial outflows, the GQA of the water is considered fair.

8. Calverley Bridge

Calverley is a rural village located near the junction of the River Aire and the Liverpool canal. It is a Holy Mill Company situated at the edge of the village. Industrial outflows make the water unsafe for use and the water is categorized as poor quality.

9. Thwaites Mill

This is a water-powered mill located on the southern edge of Leeds, used to crush stones for items such as paints and putty from 1872. As a result of crushing stones, heavy metals flow into the water, and crushed sediment increased the turbidity of water. Only tolerant species survive in this water. The Environmental Agency classify Thwaites Mill water as bad quality.

4.4 Methods

Samples of benthic invertebrates were taken from the sites in September and October of 2008 and 2009. The sites were pre-selected according to their water quality previously determined by the Environment Agency (Table 4.1 and Table 4.3). At each site, 5 replicate Surber samples ($0.25m^2$ base area, 200µm net) were taken in order to allow sample collection to be dispersed over a wide spatial area. The replicates were not used in any statistical way and the pooled samples were used for the body size analysis. The fauna collected was preserved in ethanol, identified to the lowest taxonomic level and the body size (mass) of all individuals estimated from morphometric-based formulae (Table 4.2) or, for larger individuals, by water displacement. Water quality was also assessed using invertebrate samples as the biotic index Average Score Per Taxon (Mason, 2002a) and the Biological Monitoring Working Party (BMWP) score.

BMDI

Body size distributions were plotted for each site in order to evaluate the degree of modality. In addition, individuals (and taxa) were ranked in increasing body size and the body mass difference index (BMDI) calculated between consecutive rankings using Holling's (1992) formula:

BMDI= $\mathbf{(M_{n+1}-M_{n-1})/(M_n)^{\gamma}}$

The mean BMDI was calculated as well as the mean+2SE criterion line in order to estimate the number of significant gaps in the distributions. Two consecutive differences values above the mean +2SE followed by four value below the line is considered a conservative and robust method to detect gaps (Holling, 1992). See also chapter 2.

KDE

The Kernel Density Estimation (KDE) analysis was done using the programming language 'R', in order to find the most likely number of modes (Chapter 2). The density function is the probability density function for body mass, which is produced by kernel density estimation from the distribution of body masses within the sample data and produces a smoothed version of a frequency histogram. The underlying modality of the body mass density function is determined by calculating the ‘excessive’ amount of smoothing required to produce a K-modal density function from the data as opposed to a density function with (K+1) or more. A 5% of significance level was applied to estimate the density function underlying the body mass of the benthic communities.

4.5 Results

The water quality at the sites, as determined from the ASPT and BMWP scores (Figures 4.3 and 4.4) was broadly similar to the classification provided by Environment Agency. Winterburn was the cleanest on the ASPT range and Thwaites Mill and Calverly Bridge had the poorest water quality (Figures 4.3 and 4.4). The fauna at each of these sites is shown in Table 4.2 and Figure 4.2. Winterburn was dominated by stoneflies (Leuctridae and Perlodidae), Haliplidae, Chironomidae and Simulidae. In Airton, large numbers of Haliplidae, Chironomidae, Gyrinidae, Diptera and Oligochaeta were recorded. Otterburn had many Haliplidae, Gyrinidae, Chironomidae, Diptera and Oligochaeta. The dominant taxa in Hetton were Haliplidae, Baetidae, Chiromidae, Oligochaeta and Diptera, while in Gargrave high abundances of Chironomidae, Dixidae, Haliplidae, Oligochaeta and Diptera was recorded. In Carlton Bridge, the dominant taxa were Nematomorpha, Chironomidae, Oligochaeta, Haliplidae and Baetidae. The most abundant taxa in Esholt village were Oligochaeta, Chironomidae, Hydrosychidae, Asellidae,and Nematomorpha while Calverly bridge was dominated by Oligochaeta, Chironomidae, Hydropsychidae, Nematomorpha and Asellidae. The site with the poorest water quality, Thwaites Mill, had abundant Oligochaeta, Chironomidae, Asellidae, Hydrodiidae and Viviparidae. Thus, my analyses confirm a gradient of water quality in the River Aire at these sites.

The Biological Monitoring Working Party (BMWP) score revealed the highest score of 140 and 131 for clean sites, while the lowest score (54 and 27) occurred at poor quality sites, Calverley Bridge and Thwaites Mill. The results for intermediate quality sites do not exactly match the quality gradients provided by the Environment

Agency (Figure 4.3). However, the ASPT analysis corroborated the general water quality, providing similar ranks for the sites (Figure 4.4).

Habitats at the sites

RIVPACs habitat data for the different sites on the River Aire were kindly supplied by the Environment Agency and the most revelvent variables are shown in Table 4.3. These data show that all the sites are stony shallow riffles which do not differ greatly in their substrate composition. Any between-site differences in BMWP and body size distributions are thus unlikely to be due to differences in stream bed characteristics or to bank vegetation as indicated by the shading score.

The RIVPACS data sheets also provide information on conductivity, related to dissolved solids and suspended material and general chemical characteristics of natural water (Hem, 1985). Significant changes in water conductivity could indicate pollution, pure (low conductivity) water being a good conductor of electric current. Thus, a positive relation has been found between pollution levels and conductivity (Ali *et al.*, 2009). In the present study, conductivity values were available for most sites on the Aire (Table 4.3) and there is a clear relationship between conductivity and water quality (ASPT) (Figure 4.5)

The body size distributions of benthic fauna at the sites are shown in Figure 4.6. The data here are shown on a linear body mass scale (0.25-5 mg), but the shape is similar across a range of bin sizes and transformations. All sites show a skewed distribution towards smaller size classes and none of the sites can be adequately described by a single uni-modal distribution. In addition to the left-skewed mode, there is often a clear mode around 1-2mg and possibly another mode in the largest size classes, although neither is apparent for Thwaites Mill. These modes comprise a range of taxa which differ at each site (Table 4.5).

In contrast to left-skewed distributions, analysis by Kernel Density Estimate (KDE) revealed multiple modes for the most clean site, Winterburn (Figure 4.7 Table 4.6), and a single mode for the most polluted site (Thwaites Mill). However there was no consistency in the number of modes in body size spectra for intermediate quality water sites. The sites which are considered cleaner often had fewer modes while less clean sites had more modes. Thus, the number of modes for intermediate quality water are more variable, but all sites present at least bimodality, except for the most polluted habitat.

Analysis of the number of gaps following Holling's methods shows wide variation in the number of body size gaps detected (Figure 4.9). The highest numbers of gaps occurred at Esholt, which has moderate water quality, and the lowest number of gaps was recorded in Carlton Bridge, having a fair water quality according to the Environment Agency and ASPT analysis.

There was no overall correlation between the number of modes and water quality and the number of individuals and number of modes (Figure 4.8) nor is a clear relationship between the number of gaps and water quality habitats (Figure 4.10, $R^2 = 0.0014$, $p>0.05$).

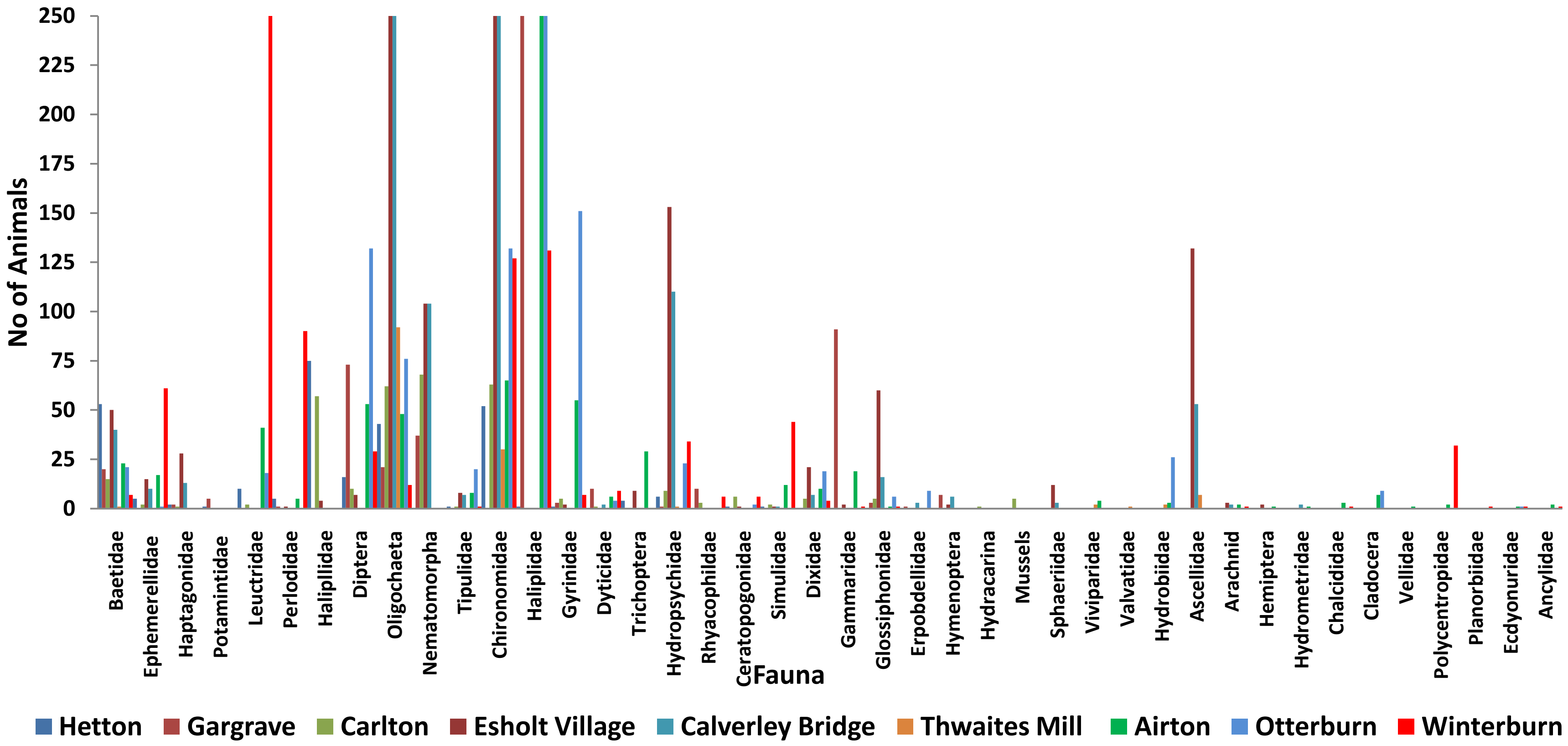
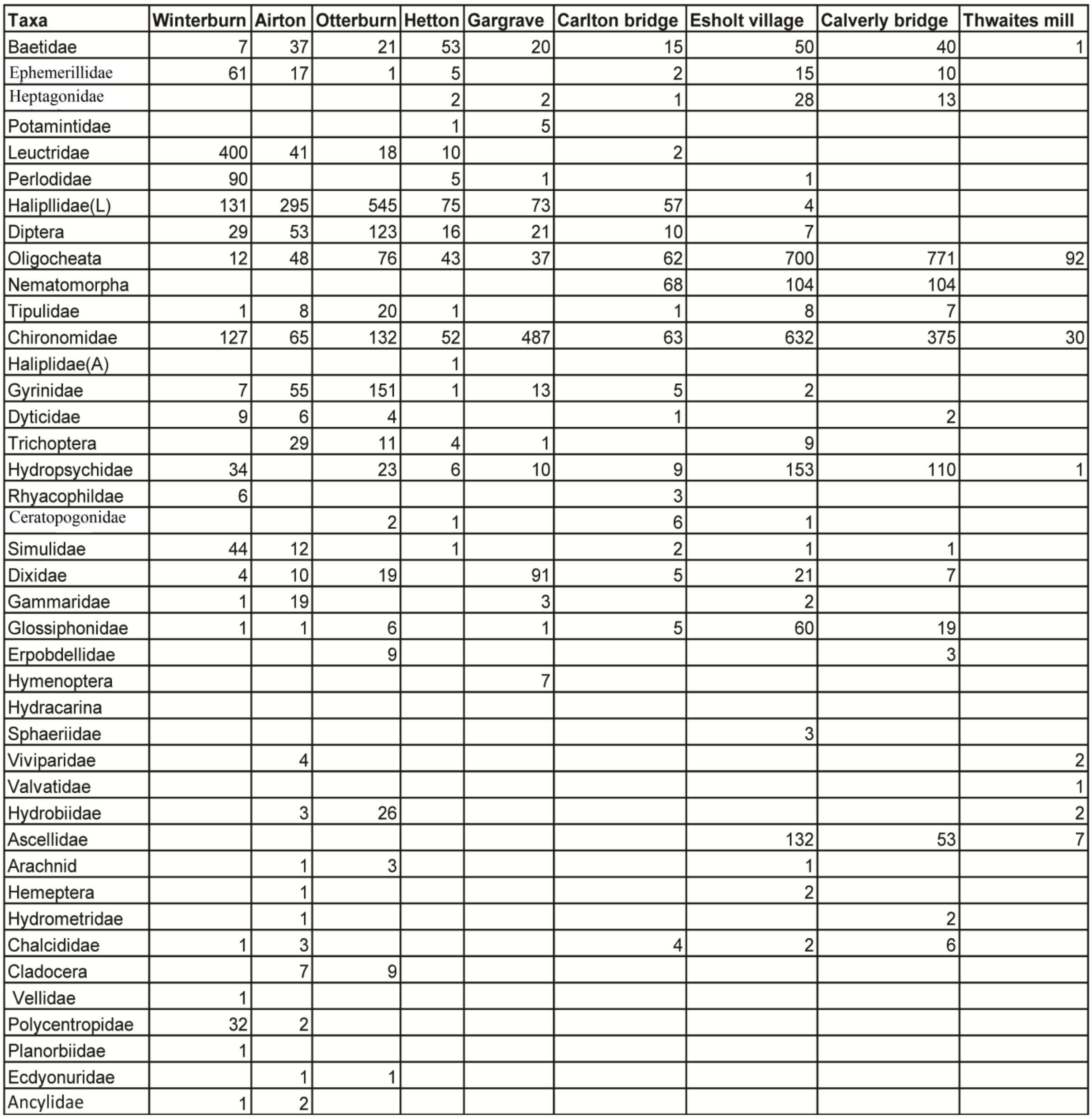


Figure 4.2. Community Index of macro invertebrates for different sites at River Aire.

Table 4.2. Community abundance of sites in the River Aire, showing the number of each taxon present at each site, where “L” is for larval stage and “A” is Adult.



Taxa	Winterburn	Airton	Otterburn	Hetton	Gargrave	Carlton bridge	Esholt village	Calverly bridge	Thwaites mill
Baetidae	7	37	21	53	20	15	50	40	1
Ephemerillidae	61	17	1	5		2	15	10	
Heptagonidae				2	2	1	28	13	
Potamintidae				1	5				
Leuctridae	400	41	18	10		2			
Perlodidae	90			5	1		1		
Halipllidae(L)	131	295	545	75	73	57	4		
Diptera	29	53	123	16	21	10	7		
Oligocheata	12	48	76	43	37	62	700	771	92
Nematomorpha						68	104	104	
Tipulidae	1	8	20	1		1	8	7	
Chironomidae	127	65	132	52	487	63	632	375	30
Haliplidae(A)				1					
Gyrinidae	7	55	151	1	13	5	2		
Dyticidae	9	6	4			1		2	
Trichoptera		29	11	4	1		9		
Hydropsychidae	34		23	6	10	9	153	110	1
Rhyacophildae	6					3			
Ceratopogonidae			2	1		6	1		
Simulidae	44	12		1		2	1	1	
Dixidae	4	10	19		91	5	21	7	
Gammaridae	1	19			3		2		
Glossiphonidae	1	1	6		1	5	60	19	
Erpobdellidae			9					3	
Hymenoptera					7				
Hydracarina									
Sphaeriidae							3		
Viviparidae		4							2
Valvatidae									1
Hydrobiidae		3	26						2
Ascellidae							132	53	7
Arachnid		1	3				1		
Hemeptera		1					2		
Hydrometridae		1						2	
Chalcididae	1	3				4	2	6	
Cladocera		7	9						
Vellidae	1								
Polycentropidae	32	2							
Planorbiidae	1								
Ecdyonuridae		1	1						
Ancylidae	1	2							

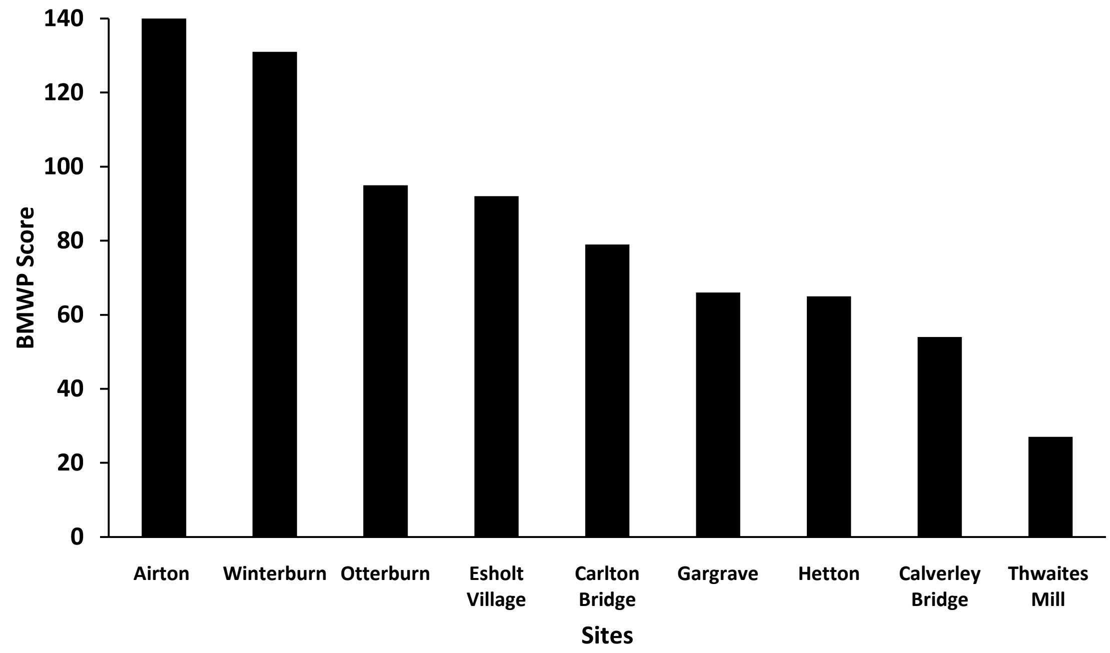


Figure 4.3. Biological Monitoring Working Party (BMWP) scores for the sites on the River Aire varying in water quality.

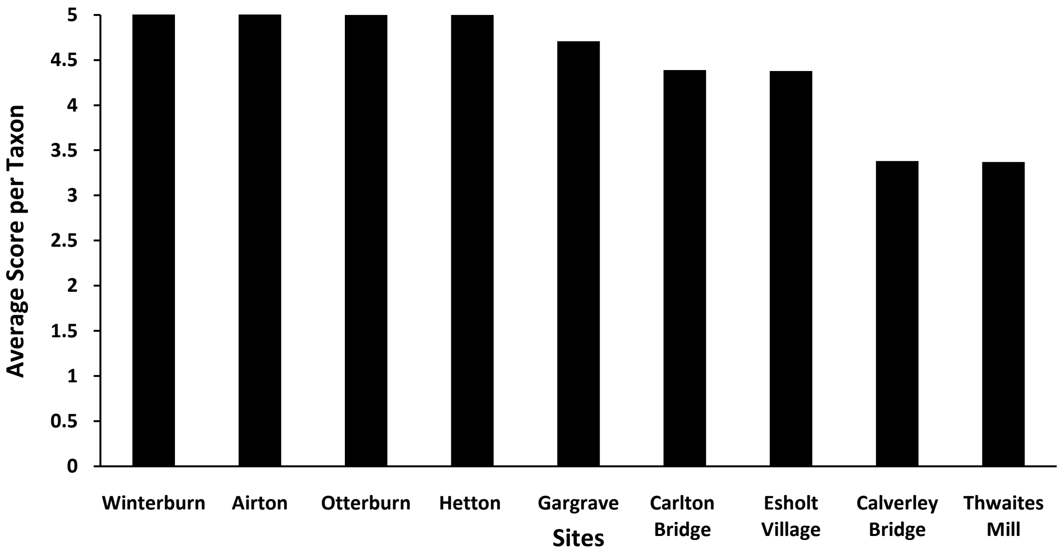


Figure 4.4. Average Score per Taxon for different sites on the River Aire, varying in water quality.

Table 4.3. RIVPACS data for different sites on River Aire provided Environmental Agency Leeds

Site	Shade	Turbidity	Odour	Width (m)	Channel depth (cm)	Conductivity (uSm)	Sewage	Bed stability	% boulders	% cobbles	% pebbles	% gravel	% sand	% silt	Riffle ?
Winterburn	Light	Slight	None	8	20	208	None	Unstable	5	65	15	5	5	0	Yes
Hetton	Moderate	Slight	None	6.2	20	275	None	Stable	10	60	15	10	5	0	Yes
Gargrave	None	Clear	None	25	25	355	None	Stable	5	60	20	10	5	0	Yes
Carlton Bridge	None	Clear	None	14	20	283	None	Unstable	0	40	40	10	10	0	Yes
Esholt Village	Light	Slight	None	15	40	381	None	Unstable	0	70	30	0	0	0	Yes
Calverley Bridge	None	Clear	None	35	30	664	None	Unstable	5	45	40	5	5	0	Yes
Thwaites Mill	None	Clear	None	20	60	348	None	Stable	5	55	30	0	10	0	Yes

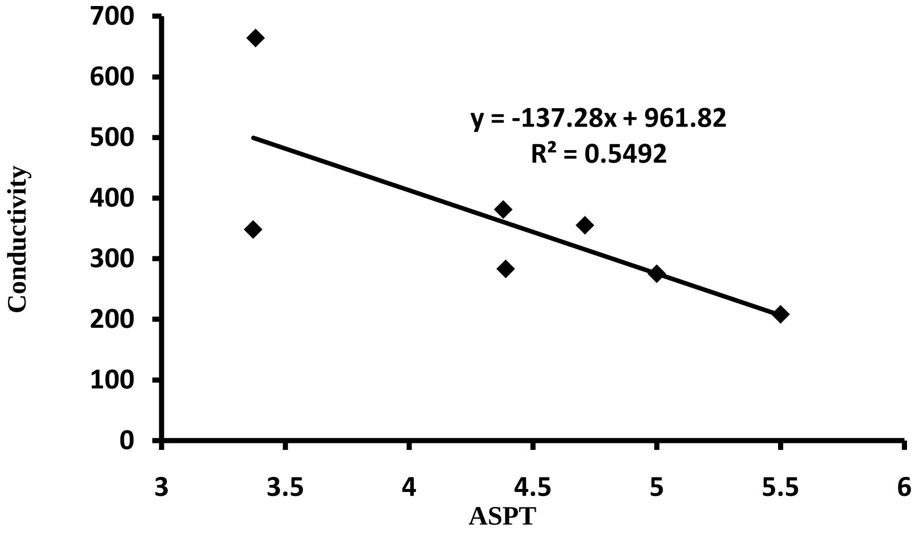


Figure 4.5. The relationship between water quality (ASPT) and water conductivity.

Table 4.4. Regression equations to determine body mass of invertebrates communities where DM is a dry mass of the organisms (mg), DW is the dry weight (mg) of the organisms, L is the length (mm) of the organisms, HW is the head width (mm) of the organisms, volume (V) of the organisms expressed in (nL) and Wt is weight of the organisms (mg).

Family/ Class	**Regression Equation To Determine Body Mass Of Fresh Water Invertebrates**	**Reference**
Baetidae Heptageniidae Caenidae Ephemerellidae Ephemeridae Potamanthidae Hymenoptera Ecdyonuridae	Dw(mg)= aL(mm)^{b} Dw(mg)=3.8x10^{-3}L(mm)$^{2.918}$	Stead *et al.*,2003
Diptera	DM(mg) = aL(mm)^{b} DM(mg) = 1.3x10^{-3} L(mm)$^{2.851}$	
Leuctriodae Perlolidae	Dw(mg)=aL(mm)^{b} DW(mg)=2.5x10^{-3}L(mm)$^{2.744}$	
Gammaridae	In DM(mg)=Ina +b InL(mm) In DM(mg)=-4.95 +2.83 InL(mm)	
Tipulidae	DW(mg) = aL(mm)$^{2.851}$ Dw(mg)= 1.3x10^{-3} L(mm)$^{2.851}$	
Chironomidae Caratopogonidae	DM (mg) = a L(mm)^{b} DM (mg) = 6.0x 10^{-4} L (mm) $^{2.770}$	
Hemerobiidae	Log DM(µg)=a + b log HW(mm) Log DM(µg)=2.68+2.9 log Hw(mm)	
Trichoptera Rhyacophilidae Hydropsychidae	In DM(mg)=Ina + b InL(mm) In DM (mg) = -6.037 + 2.82 In L (mm).	
Simulidae	In DM(mg)=Ina + b Hw(mm) In DM(mg)= -4.5009 +2.0742 HW(mm)	
Arachinida Argulidae	DM(µg) = aL(µm)^{b} DM(µg) = 1.1x10^{-5}L(µm)$^{1.89}$	
Oligochaeta	DM(nl) =a L(µm)^{b} DM(nl) = 3.5 x10^{-3} L(µm)$^{2.1}$	
Dixidae	DM(mg)=aL (µm)^{b} DM(mg)=6.62x10^{-4} L(µm)$^{2.59}$	
Cladocera	InDM(µg) = Ina+bInL(mm) InDM(µg) = In1.7512+ 2.653L(mm)	
Asellidae	DM(mg) = aL(mm)^{b} DM(mg) = 7.2x10^{-3}L(mm) $^{2.785}$	
Nematomorpha	DM(µg)=a L(µm)^{b} DM(µg) =6.0x10^{-5} L(µm)$^{0.8205}$	

Turbullaria	V(nL)=L(mm)xW^2mm) x C V(nL)=L(mm)W^2(mm)x 550 V(nL) ×1.05 = dry weight= *µg* *µg/1000=mg*	(Feller and Warwick, 1988)
Piscicolidae Erpobdellidae Glossiphonidae	V(nL) = L(mm)×π$(W/2)^2$ ×530 V(nL)×1.13 = dry wight(µg)	
Valvatidae Uniondae Planorbidae	**Water Displacement**	(Leaper *et al.*, 2001)
	V(nL)=WD(µL)x 1000 Wt(µg)=v(nLx1.05) Mass(mg)=µg/1000	
Hydrobiidae Physidae Viviparidae	V(µL) = L(mm) $(0.851)^{1.91}$ Wt(µg) = v(µLx 1.05) Mass(mg) = µg/1000	
Ancylidae Sphaeriidae	**Approximate a geometric shape (cone)**	
	V(µL) = 1/3πr^2 (mm) h(mm) V(nL)= µLx1000 Wt(µg) = nLx1.05 Mass (mg) = µg/1000	
Corixidae Notonectidae Mesovellidae Veliidae Hydrometridae Hemiptera	In W(mg) = Ina + b In L In W(mg) = -4.200+ 2.60 In L(mm)	(Smock, 1980)
Gyrinida Dyticidae Haliplidae	Dw (mg)= Ina + b In L(BL(mm) or HW(mm) Dw = -2.0076 +3.2271 InL(BL-Dw) Dw =3.1102 +2.5412 InL(HW-DW) converted BL to HW by using HW:BL	(Tower *et al.*, 1994)

Taxa	**Hetton**	**Gargrave**	**Carlton Bridge**	**Esholt Village**	**Calverley**	**Thwaites Mill**	**Airton**	**Otterbun**	**Winterbun**
Baetidae	0.00445-0.734	0.00973-1.07	0.065-2.5	0.012-0.9	0.02-0.91	0.04	0.03-2.6	0.0011-7.1	0.05-0.2
Ephemerellidae	0.073-0.62		0.01-0.14	0.02-1.3	0.04-0.73		0.1-1.92	0.07	0.003-0.9
Heptageniidae	0.04-0.074	0.13-0.6	0.012	0.03-0.91	0.04-0.87				
Potaminthidae	0.1702	0.0432-0.6707							
Leuctridae	0.011-0.109		0.6-0.8				0.0054-0.8	0.13-1.2	0.009-0.92
Perlodidae	0.0832-0.1309	0.578		0.05					0.006-1.031
Haliplidae(L)	0.196-4.505	2.27-4.27	0.42-3.506	0.7-4.2			0.2-5.6	0.2-5.3	0.2-4.45
Diptera	0.03-0.89	0.213-2.78	0.04-2.4	0.03-1.8			0.04-2.4	0.0092-1.9	0.003-2.2
Oligochaeta	0.04-1.72	0.00689-2.33	0.0043-3.72	0.003-3.1	0.013-2.98	0.007-1.99	0.03-10.96	0.02-5.9	0.2-7.7
Nematomorpha			0.00003-0.00015	0.00003-0.0002	0.00003-0.0002				
Tipulidae	0.74		0.33	0.13-3.1	0.0085-0.05		0.3-2.7	0.0092-6.5	0.5
Chironomidae	0.0037-0.146	0.00062-0.45	0.0005-0.2095	0.0005-0.132	0.0007-0.054	0.003-0.09	0.0007-0.3	0.0007-0.231	0.0013-0.7
Haliplidae(A)	0.012								
Gyrinidae	0.012	0.12-2.197	0.12-0.4112	0.12-0.3			0.2-2.5	0.2-2.94	0.2-2.5
Dytiscidae			0.12		0.3-0.54		0.6-1.98	0.2-1.9	0.2-1.13
Trichoptera	0.069-0.81	0.18		0.02-0.92			0.005-2.6	0.07-6.5	
Hydropsychidae	0.032-0.45	0.00843-2.6	0.0134-1.84	0.012-7	0.004-7.6	0.2		0.12-5.13	0.07-3.5
Rhyacophilidae			0.001-0.2						0.4-1.54
Caratopogonidae	0.08504		0.072-0.22	0.07				0.042-0.082	
Simuliidae	0.02		0.02-0.03	0.02	0.02		0.01-0.04		0.02-0.044
Dixidae		0.00143-0.0246	0.57-1.63	0.3-1.622	0.0054-0.011		0.004-0.007	0.004-0.022	0.06-0.132
Gammaridae		0.017-0.193		0.06-0.13			0.11-3.1		2.5
Glossiphonidae		41.3	0.3-45.8	0.81-30.9	1.72-29.4		22.94	3.93-9.74	17.32
Erpobdellidae					0.53-1.5			15.6-73.6	
Hymenoptera		0.014-0.2							
Sphaeriidae				0.02-0.09					
Viviparidae						0.003-0.004	0.003-0.0041		
Valvatidae						0.2			
Hydrobiidae						2	0.0022-0.0041	0.002-0.0041	
Asellidae				0.03-1.9	0.091-2.002	0.12-2.22			
Arachnida				0.0000002				0.000000005-0.0000002	
Hemiptera				1.1-2.03			0.24		
Hydrometridae					2		0.02		
Chalcididae			0.2-0.3	0.03-0.039	6		0.0045-0.07		0.09
Cladocera							0.005-0.006	0.005-0.007	
Veliidae							0.02		0.015
Polycentropidae							0.34-1.54		0.05-3.5
Planorbiidae									1.1
Ecdyonuridae							2.02	0.03	
Ancylidae							0.006-0.007		0.02

Table 4.5. Size ranges of all taxa recorded in the present study, where a range is not shown the data is referring to single individual

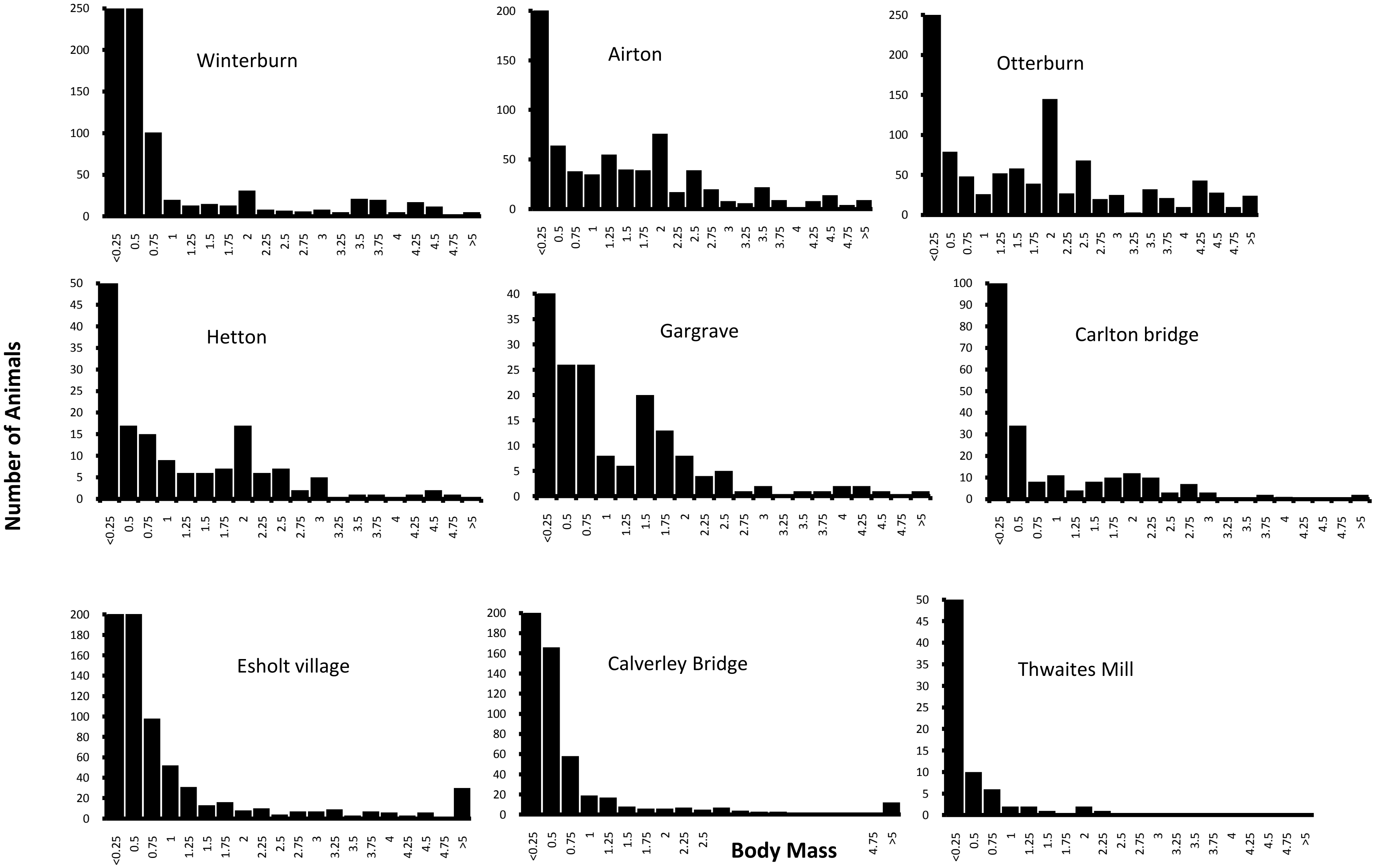


Figure 4.6. Body size distributions for invertebrates in the River Aire flowing from Winterburn to Thwaites Mill. Body masses were measured in mg, ranging from <0.02 to >5.

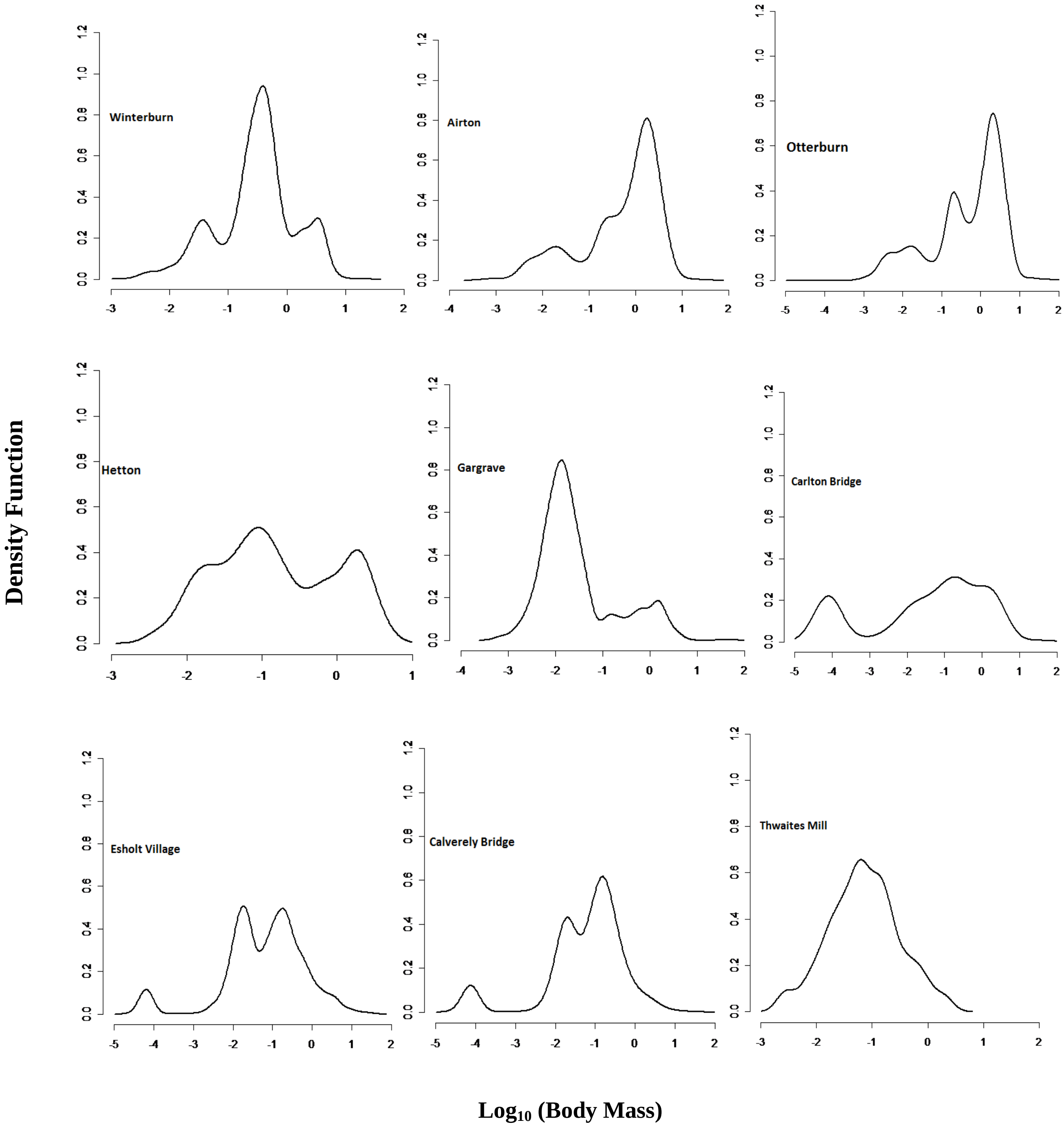
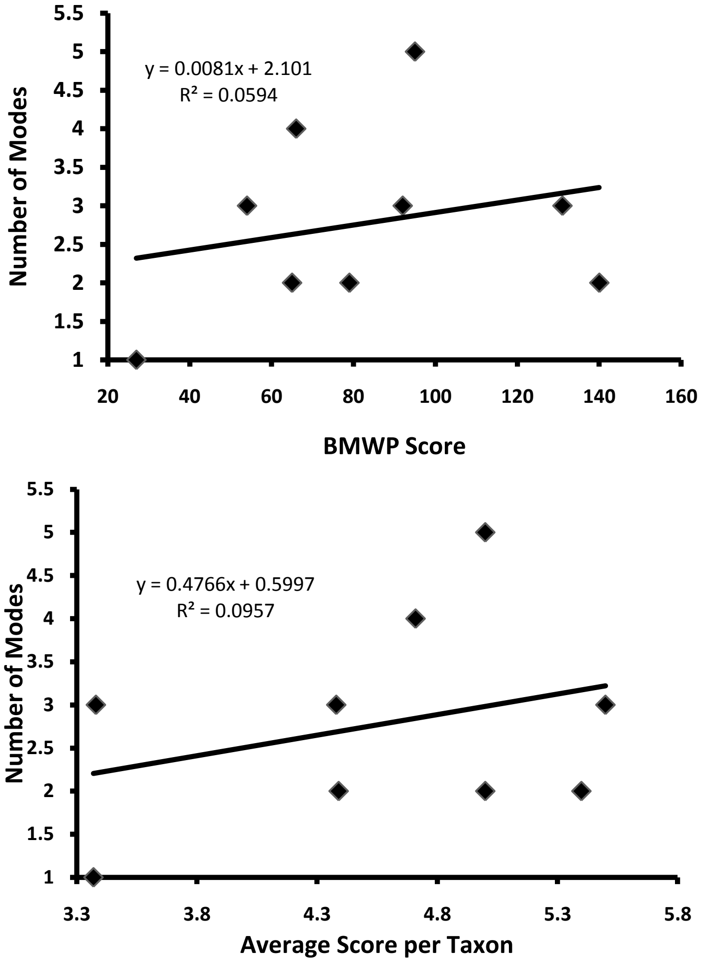


Figure 4.7. The fitted distributions for density-body size estimated by Kernel Density Estimation and bootstrapped re-sampling in the River Aire. Both axes are scaled as $\log_{10}$ of the original data. Density function = number of individuals at each site.

Table 4.6. Results of the test of significance for the body size spectra from the kernel density estimation and smoothed bootstrap re-sampling, where h is a smoothing constant (critical bandwidth) used in kernel estimation, m is the smallest number of modes for which the bootstrap test was not significant at the 5% level, and α is the level of significance for each distribution (mode number).

Sites	**h**	**m**	**α**
Winterburn	0.126	3	0.496
Airton	0.178	2	0.292
Otterburn	0.178	5	0.182
Hetton	0.173	2	0.338
Gargrave	0.137	4	0.095
Carlton Bridge	0.353	2	0.103
Esholt Village	0.128	3	0.467
Calverly Bridge	0.194	3	0.055
Thwaites mill	0.153	1	0.802



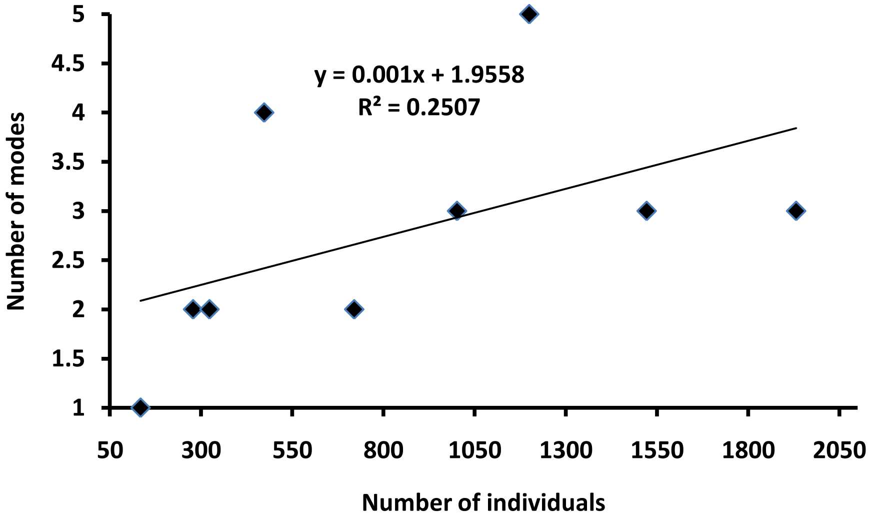


Figure 4.8. The relationships between water quality assessed by biotic indices (ASPT, BMWP scores and number of individuals) and number of modes by KDE in body mass distributions at different sites of River Aire. Both relationships are not statistically significant.

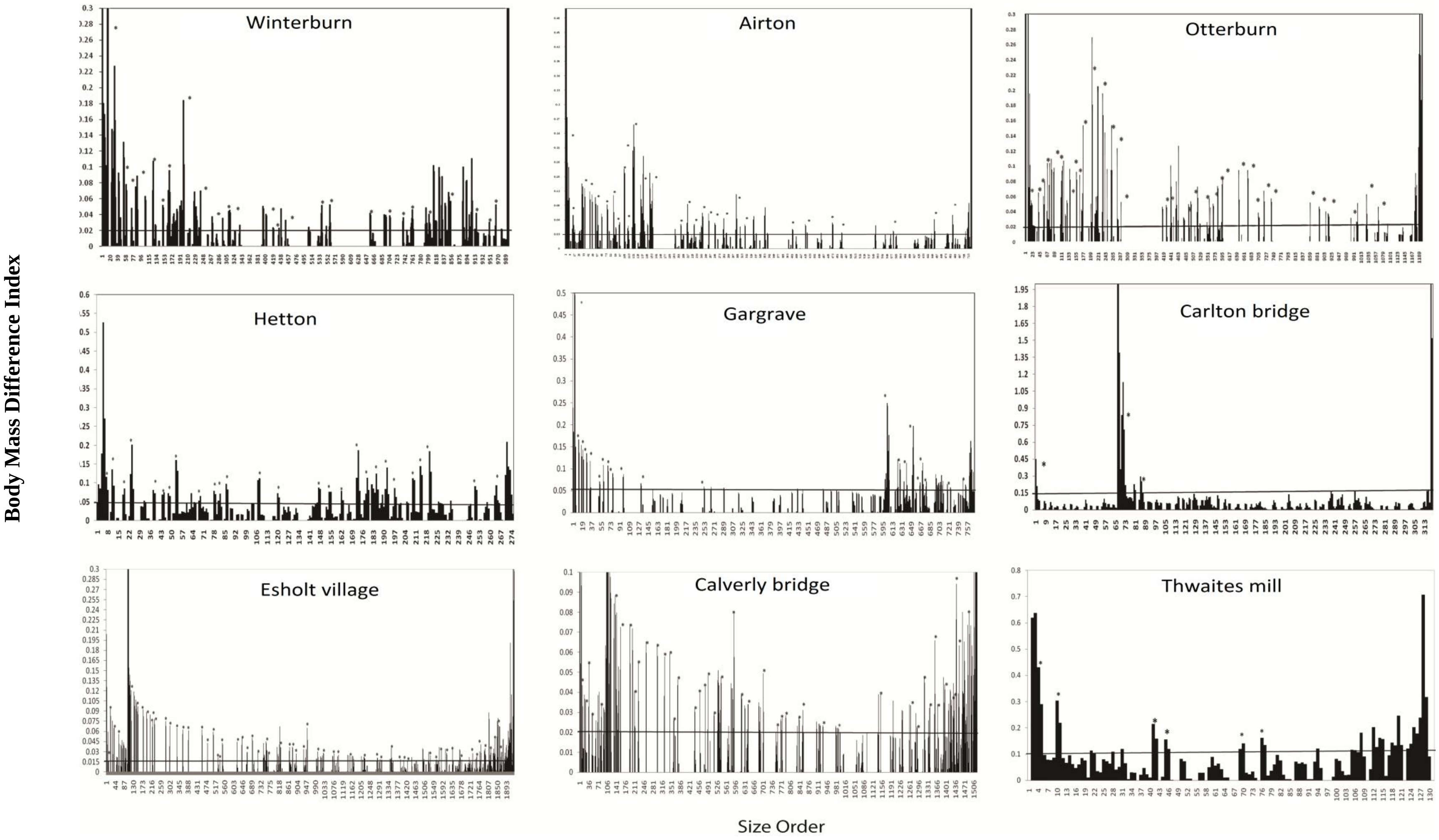


Figure 4.9. Distributions of body mass gaps for river Aire stream communities. The horizontal line is the mean +2SE and asterisks (*) show significant body mass gaps, identified as at least two values of BMDI.

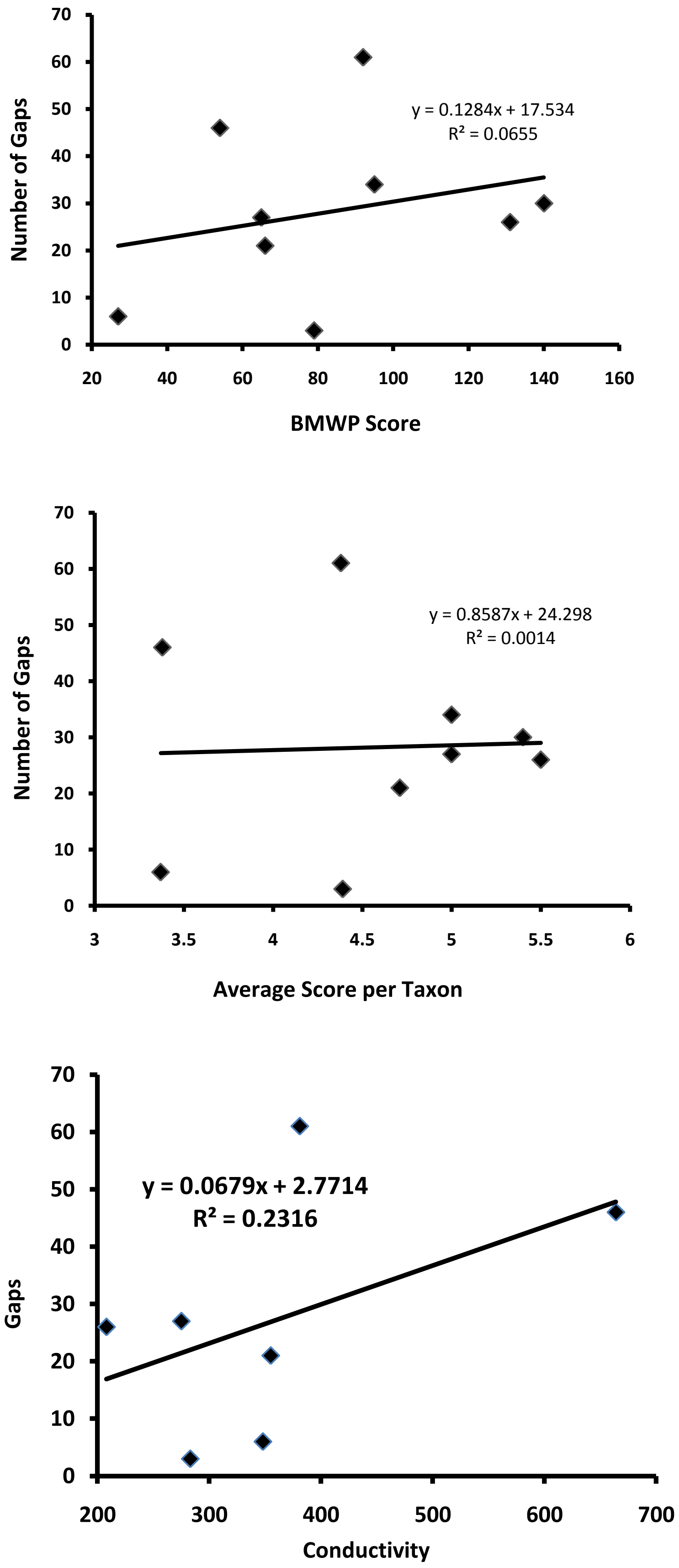
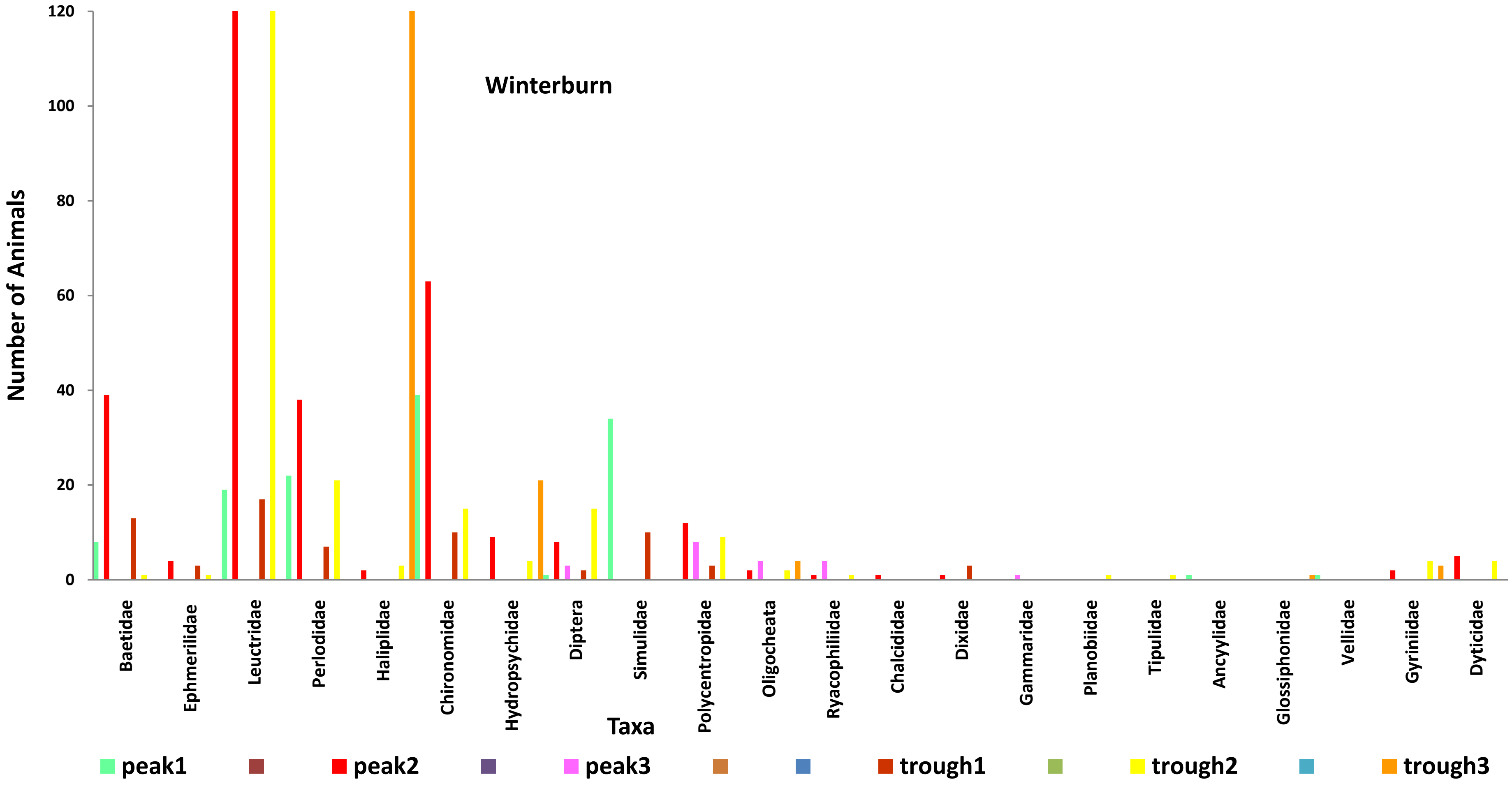
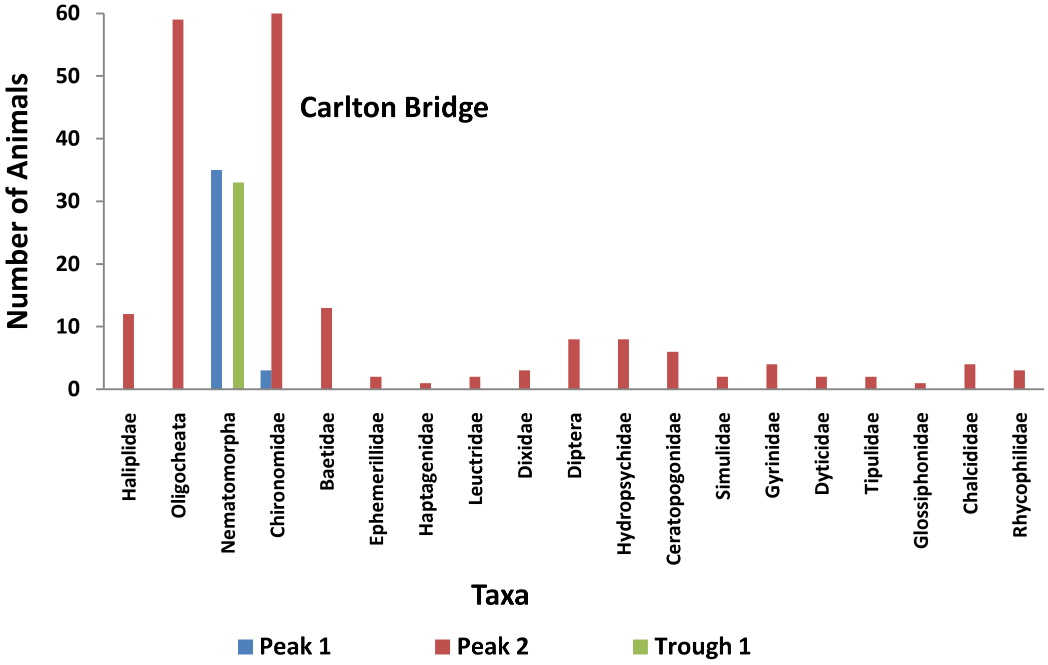


Figure 4.10. The relationship between water quality determined by biotic indices (BMWP, ASPT and conductivity) and number of gaps detected in body mass distributions at different sites of River Aire. All relationships are not significant.

The data were explored further for phylogenetic effects, a potential explanation of modes occurring at particular body sizes.(Figure 4.11). Thus, the peak and troughs in the body mass distributions do not appear to characterised by particular taxa, being a mixture of different invertebrate groups. This can be seen in the figures below by comparing where the different groups (individually coloured) are in the size plots: clearly, groups do not always appear in the same place on the x-axis.This suggests that abundance was highest at the clean sites and that taxa are not aggregated at specific locations (peaks and troughs), being more evenly distributed in their body sizes. Thus, the peak and troughs in the body mass distributions do not appear to characterised by particular taxa, being a mixture of different invertebrate groups. Phylogenetic explanations for the number and locations of peaks and troughs therefore seem unlikely.


Winterburn
Number of Animals
120
100
80
60
40
20
0
Baetidae
Ephmerilidae
Leuctridae
Perlodidae
Haliplidae
Chironomidae
Hydropsychidae
Diptera
Simulidae
Polycentropidae
Oligocheata
Ryacophiliidae
Chalcididae
Dixidae
Gammaridae
Planobiidae
Tipulidae
Ancyylidae
Glossiphonidae
Vellidae
Gyriniidae
Dyticidae
Taxa
peak1
peak2
peak3
trough1
trough2
trough3



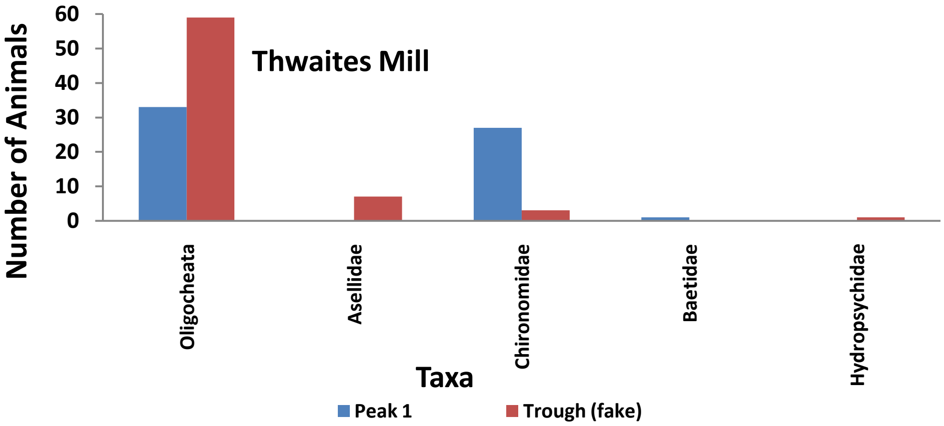


Figure 4.11 The occurenece of taxa in peaks and troughs are presented with different colours in clean, intermediate and polluted sites in River Aire. Number of colours showed in labels indicating number of taxa present in peaks or trough.

4.6 Discussion

This study was carried out to assess the effects of water quality on body mass distributions in stream communities. Benthic fauna were sampled across different sites on the River Aire varying in water quality, to establish the potential of size-based indicators for assessing environmental conditions. Water quality had been classified by the Environment Agency and also assessed in this study using invertebrate samples as the BMWP score and the Average Score per Taxon (Mason, 2002). The ASPT estimates for the sites were similar to the classification provided by Environment Agency. The highest score of 5.5 was for the cleanest site and scores decreased with decreased water quality, with the lowest score of 3.37 for the most polluted sites. The RIVPACs data for the sites provided by the Environmental Agency (Table 4.3) show that habitat difference between the sites is minimal: all the sites are riffles with no silt. Thus the main difference between the sites is in their water quality.

Looking at the histograms, most sites were clearly not unimodal with respect to their body size distributions. Many authors have claimed that body mass distributions in communities are multimodal (Schwinghamer, 1981a; Poff *et al.*, 1993), although some do show uni-modal size spectra (Solimini *et al.*, 2001). In the River Aire, visual inspection of body size distributions showed at least bimodality for invertebrates, mainly in the cleaner sites which supports a wide range of body sizes. The most polluted site (Calverly Bridge and Thwaites Mill) were better described as unimodal. There was a large number of small size individuals within these sites.

The modality in frequency histograms is considered sensitive to bin width. If the bin is too wide, important information may be missed, while a too narrow bin width gives un-meaningful information. Choosing inappropriate bin-widths therefore may create errors in interpretation of histogram (Denby and Mallows, 2009). The body mass patterns analysed using Kernel Density Estimates show multimodality for most clean sites, e.g. Otterburn had 5 modes and Winterburn had 3 modes, while a single mode was recorded for most polluted site (Table 4.6). The intermediate water quality sites show variable numbers of modes in their body size distributions, the fair quality habitat Gargrave having 4 modes followed by fairly good and poor quality sites (Esholt village and Calverley Bridge) with 3 modes. Thus, there was no overall correlation between the number of modes and water quality (Figure 4.8). The effect of sample size on body mass distributions examined. This showed that increase in sample size is unlikely change the estimate of scaling exponent for body mass distribution (White and Seymour, 2005). In present study there was no relationship between number of individuals and number of modes in body mass distributions. (Figure 4.8)

Whilst the above presents some evidence for changes in modality with water quality, this was not reflected in the BMDI analysis. For gap analysis, more discontinuities (body size gaps) imply less disturbed communities (Holling, 1992), but in my study there was no relationship between the number of gaps and water quality as measured by ASPT and BMWP estimates. Many gaps were observed at Esholt village which had a fairly good quality compared to the cleaner site (Winterburn). 61 gaps were recorded in Esholt village followed by Calverly Bridge with 46 gaps. Otterburn, Airton and Hetton are good quality habitats but in gap analysis these sites revealed 34, 30 and 27 gaps, respectively. At the cleanest site (Winterburn), the number of gaps was 26 while in Gargrave 21 gaps were recorded. One of the lowest numbers of gaps was recorded in the polluted site (Thwaites Mill) with 6 gaps, but, in contrast, a fair quality habitat (Carlton Bridge) had only 3 gaps. Thus, overall there was no clear relationship between water quality (ASPT and conductivity) and the number of gaps (Figure 4.11). Finding gaps using the BMDI approach seems to be sensitive to the presence of exceptionally large values of BMDI that increase the mean value hugely making it almost impossible to detect the gaps which occur amongst the lower BMDI values.

In the introduction the leading competing hypotheses explaining body size distributions in ecosystems were listed. The appropriate scale varies for each hypothesis and there is no evidence that one scale is superior to other scales for analysis (Vermaat *et al.*, 2005) because different processes are important at different scales, and so no single theory might explain the patterns across different scales (Gaston *et al.*, 2001). To link body mass patterns to the processes affecting those patterns, multiscale analysis is critical (Krawchuk and Taylor, 2003), but there is evidence of multimodality (Havlicek and Carpenter, 2001) and discontinuity in body mass distribution in a range of ecosystems (Allen *et al.*, 1999). The present study compares adjacent systems with different environmental conditions, so that the taxonomic identities of species and their evolutionary histories will be similar, and thus phylogenetics are held constant. The system is spatially connected and allows species to disperse across the habitats which varies from clean to polluted and the different body mass patterns observed cannot be due to biogeographical separation. The community interaction hypothesis predicts that changes in the patterns of body size are because of different taxa present in the system, but such taxonomic differences are restricted to the species level, not the higher-level taxa dealt with here. The textural discontinuity hypothesis predicts changes in body size patterns because the habitat available to the animals differs. The presence of multiple modes and gaps in cleaner sites reflects the hierarchical physical structure of the system and shows that multiple processes are responsible for structuring the system. At the most polluted site (Thwaites Mill), fewer modes might be a sign of a disturbed and less resilient system, although the number of gaps for other polluted sites is higher and for some clean sites is lower.

In other studies (not freshwater streams), multiple modes in body mass distributions have been considered as a result of faunas with different macro-evolutionary histories (Cassey and Blackburn, 2004). Studies on South American mammals have revealed that body mass distributions are made of different faunal stocks with different macro-evolutionary histories (Marquet and Cofre, 1999). In the present study this phylogenetic explanation is unlikely to explain the multiple modes in body mass distributions. Figure 4.11 revealed that the number modes decreased with water quality but the peaks and troughs in body mass distributions for clean, intermediate and poor quality water contains the body masses of all animals which comprise a mixture of different groups and specific groups are spread across peaks and troughs. It is noted that the same species (family) are not accumulated in specific peaks or troughs while the body mass distributions within modes are mixture of all.

In conclusion, it is clear that body size distributions in stream communities are "lumpy", in the sense that most sites show more than one mode or many gaps. The most polluted site in our study had the fewest modes and gaps and the cleanest site had many gaps and was clearly multimodal. In terms of Holling's (1992) Textural Discontinuity Hypothesis, these patterns could reflect the dynamic processes operating at particular scales, no other competing hypotheses seeming plausible. However, it is also clear that the number of gaps (discontinuities) is not well correlated with disturbance, at least with water quality, and it is difficult to demonstrate such associations at smaller scales (Leaper *et al.*, 2001). Further empirical exploration of such relationships is needed given their compelling theoretical basis (Holling, 1992).

4.7 References

Ali, N., Ahmed, E., Othman, A. K. and Othman, J. (2009). Prediction of johor river water quality parameters using artificial neural networks. *European Journal of Scientific Research* 28(3): 422-435.

Allen, C., Gunderson, L. and Johnson, A. R. (2005). The use of discontinuities and functional groups to assess relative resilience in complex systems. *Ecosystems* 8(8): 958-966.

Allen, C. R., Forys, E. A. and Holling, C. S. (1999). Body mass patterns predict invasions and extinctions in transforming landscapes. *Ecosystems* 2(2): 114-121.

Allen, C. R., Garmestani, A. S., Havlicek, T. D., Marquet, P. A., Peterson, G. D., Restrepo, C., Stow, C. A. and Weeks, B. E. (2006). Patterns in body mass distributions: Sifting among alternative hypotheses. *Ecology Letters* 9(5): 630-643.

Blackburn, T. M. and Gaston, K. J. (1994). Animal body size distributions: Patterns, mechanisms and implications. *Trends in Ecology & Evolution* 9(12): 471-474.

Blackburn, T. M. and Gaston, K. J. (1999).The relationship between animal abundance and body size: A review of the mechanisms. In *Advances in ecological research*, Vol. Volume 28, 181-210 (Eds A. H. Fitter and D. Raffaelli). Academic Press.

Brown, J. H., Marquet, P. A. and Taper, M. L. (1993). Evolution of body size: Consequences of an energetic definition of fitness. *The American Naturalist* 142(4): 573.

Cassey, P. and Blackburn, T. M. (2004). Body size trends in a holocene island bird assemblage. *Ecography* 27(1): 59-67.

Cyr, H., Peters, R. H. and Downing, J. A. (1997). Population density and community size structure: Comparison of aquatic and terrestrial systems. *Oikos* 80(1): 139-149.

Damuth, J. D. (1992). Taxon- free characterization of animal communities. . *In:Terrestrial Ecosystems Through Time Evolutionary Paleoecology of Terrestial plants and Animals(eds Behrensmeyer, A.K., Damuth,J.D., DiMichele,W.A., Potts,R.,Sues,H&Wing,S.L.).University of Chicago Press,Chicago,IL,USA.* : 183 -203.

Denby, L. and Mallows, C. (2009). Variations on the histogram. *Journal of Computational and Graphical Statistics* 18(1): 21-31.

Fu, C., Wu, J., Wang, X., Lei, G. and Chen, J. (2004). Patterns of diversity, altitudinal range and body size among freshwater fishes in the yangtze river basin, china. *Global Ecology & Biogeography* 13(6): 543-552.

Gardezi, T. and Silva, J. d. (1999). Diversity in relation to body size in mammals: A comparative study. *The American Naturalist* 153(1): 110-123.

Gaston, K. J. and Blackburn, T. M. (1996). Range size-body size relationships: Evidence of scale dependence. *Oikos* 75(3): 479-485.

Gaston, K. J., Chown, S. L. and Mercer, R. D. (2001). The animal species-body size distribution of marion island. *Proceedings of the National Academy of Sciences of the United States of America* 98(25): 14493-14496.

Havlicek, T. D. and Carpenter, S. R. (2001). Pelagic species size distributions in lakes: Are they discontinuous? *Limnology and Oceanography* 46(5): 1021-1033.

Hem, J. D. (Ed) (1985). *Study and interpretation of the chemical characteristics of natural water.*: US Govt Printing Office, Washington.

Holling, C. S. (1992). Cross-scale morphology, geometry, and dynamics of ecosystems. *Ecological Monographs* 62(4): 447-502.

Hutchinson, G. E. (1959). Homage to santa rosalia, or why are there so many kinds of animals? *Am.Nat.,* 93: 245-249.

Krawchuk, M. A. and Taylor, P. D. (2003). Changing importance of habitat structure across multiple spatial scales for three species of insects. *Oikos* (103): 153-161.

Leaper, R., Raffaelli, D., Emes, C. and Manly, B. (2001). Constraints on body-size distributions: An experimental test of the habitat architecture hypothesis. *Journal of Animal Ecology* 70(2): 248-259.

Lovegrove, B. G. and Haines, L. (2004). The evolution of placental mammal body sizes: Evolutionary history, form, and function. *Oecologia* 138(1): 13-27.

Marquet, P. A. and Cofre, H. (1999). Large temporal and spatial scales in the structure of mammalian assemblages in south america: A macroecological approach. *Oikos* 85: 299–309. .

Marquet, P. A., Navarrete, S. A. and Castilla, J. C. (1995). Body size, population density, and the energetic equivalence rule. *Journal of Animal Ecology* 64(3): 325-332.

Mason, C. F. (Ed) (2002). *Biology of fresh water pollution.*: Pearson Educated Limited.

Maurer, B. A. and Brown, J. H. (1988). Distribution of energy use and biomass among species of north american terrestrial birds. *Ecology* 69(6): 1923.

Morse, D. R., Stork, N. E. and Lawton, J. H. (1988). Species number, species abundance and body length relationships of arboreal beetles in bornean lowland rain forest trees. *Ecological Entomology* 13(1): 25-37.

Poff, N. L., Palmer, M. A., Angermeier, P. L., Vadas, R. L., Jr., Hakenkamp, C. C., Bely, A., Arensburger, P. and Martin, A. P. (1993). Size structure of the metazoan community in a piedmont stream. *Oecologia* 95(2): 202-209.

Pyron, M. (1999). Relationships between geographical range size, body size, local abundance, and habitat breadth in north american suckers and sunfishes. *Journal of Biogeography* 26(3): 549-558.

Schwinghamer, P. (1981). Characteristics size distribution of integrakl benthic communities. *Candian Journal of Fishries and Aquatic Science* 38: 1255-1263.

Schwinghamer, P. (1988). Influence of pollution along a natural gradient and in mesocosm experiment on biomass- size spectra of benthic communities. *Mar. Ecol- Prog Ser.* 46:: 199-206.

Sheldon, R. W., A.Prakash and W.H.Sutcliffe, J. (1972). The size distribution of particles in the oceaon. *Limnology and Oceanography* 17:: 327-340.

Silva, M., Brimacombe, M. and Downing, J. A. (2001). Effects of body mass, climate, geography, and census area on population density of terrestrial mammals. *Global Ecology and Biogeography* 10(5): 469-485.

Smith, F. A., Brown, J. H., Haskell, J. P., Lyons, S. K., Alroy, J., Charnov, E. L., Dayan, T., Enquist, B. J., Ernest, S. K. M., Hadly, E. A., Jones, K. E., Kaufman, D. M., Marquet, P. A., Maurer, B. A., Niklas, K. J., Porter, W. P., Tiffney, B. and Willig, M. R. (2004). Similarity of mammalian body size across the taxonomic hierarchy and across space and time. *The American Naturalist* 163(5): 672-691.

Solimini, A. G., Benvenuti, A., D'Olimpio, R., Cicco, M. D. and Carchini, G. (2001). Size structure of benthic invertebrate assemblages in a mediterranean river. *Journal of the North American Benthological Society* 20(3): 421-431.

Thiebaux, M. L. and Dickie, L. M. (1992). Models of aquatic biomass size spectra and the common structure of their solutions. *Journal of Theoretical Biology* 159(2): 147-161.

Vermaat, J. E., Eppink, F., van den Bergh, J. C. J. M., Barendregt, A. and van Belle, J. (2005). Aggregation and the matching of scales in spatial economics and the matching of scales in spatial economics and landscape ecology:Empirical evidence and prospects for integration. *Ecological Economics* 52(2): 229-237.

White, C. R. and Seymour, R. S. (2005). Sample size and mass range effects on the allometric exponent of basal metabolic rate. *Comparative Biochemistry and Physiology - Part A: Molecular & Integrative Physiology* 142(1): 74-78.

Chapter 5

Body mass distributions of benthic communities in static water habitats (canals and ponds)

5.1 Introduction

The macro-invertebrate community in the littoral zone of static bodies like ponds, canals and lakes are important in transferring energy from primary producers and detritus to fish (Boisclair and Leggett, 1985). Several factors influence their distributions including macrophytes (James *et al.,* 1998), and, in canals, geomorphological processes influence the groundwater circulation at the hyporheic layer of the canal system (Chatelliers and Reygrobellet, 1990). Cosser (1989) found that the distributions of macro-invertebrates communities were highly connected with water quality and dissolved oxygen in the canal systems of the Gold Coast, Queensland, Australia. Body size distributions changed in benthic communities in response to nutrients in freshwater ponds (Hall *et al.,* 1970) and in lakes there are associations with aquatic plants, which provide shelter from fish predation (Hanson, 1990). Generally, the biomass-body size scaling relationship of lentic macro-invertebrates seems to be controlled by the fractal dimension of a habitat and the allometric scaling of resources (McAbendroth *et al.*, 2005).

However, relatively little information is available on the size structure of freshwater benthic communities (Strayer, 1986; Hanson, 1990). Differences occur among lakes in macrobenthic size structure, influenced by factors such as water chemistry, lake productivity, sediment compositions and vertebrate predation (Rath, 1986; Strayer, 1991). Quantitative descriptions of size structure revealed unimodal distributions for the benthic animal community in Mirror Lake, New Hampshire (Strayer, 1986), while Rasmussen (1993) claimed bimodal distributions of body mass in 11 lakes of the Quebec Eastern Township. The size spectra seem to be quite variable in many studies of lentic macro-invertebrate communities (Hanson *et al.*, 1989b; Morin and Nadon, 1991). However, most of the above studies were not carried out in the context of resilience, or using rigorous approaches for determining multimodality or gaps in size distributions, such as the KDE and BMDI methods used chapters 3 and 4, where the size distributions in streams communities (Chapter 3) were found to be ‘lumpy’ and associations have been found between modality and water quality. The question therefore arises as to whether such patterns also occur in static water bodies. In this chapter, the same relationships are explored for static water bodies, specifically I analyse the body mass distributions of macro-invertebrates in a range of static water bodies which vary in their water quality for modality and number of gaps.

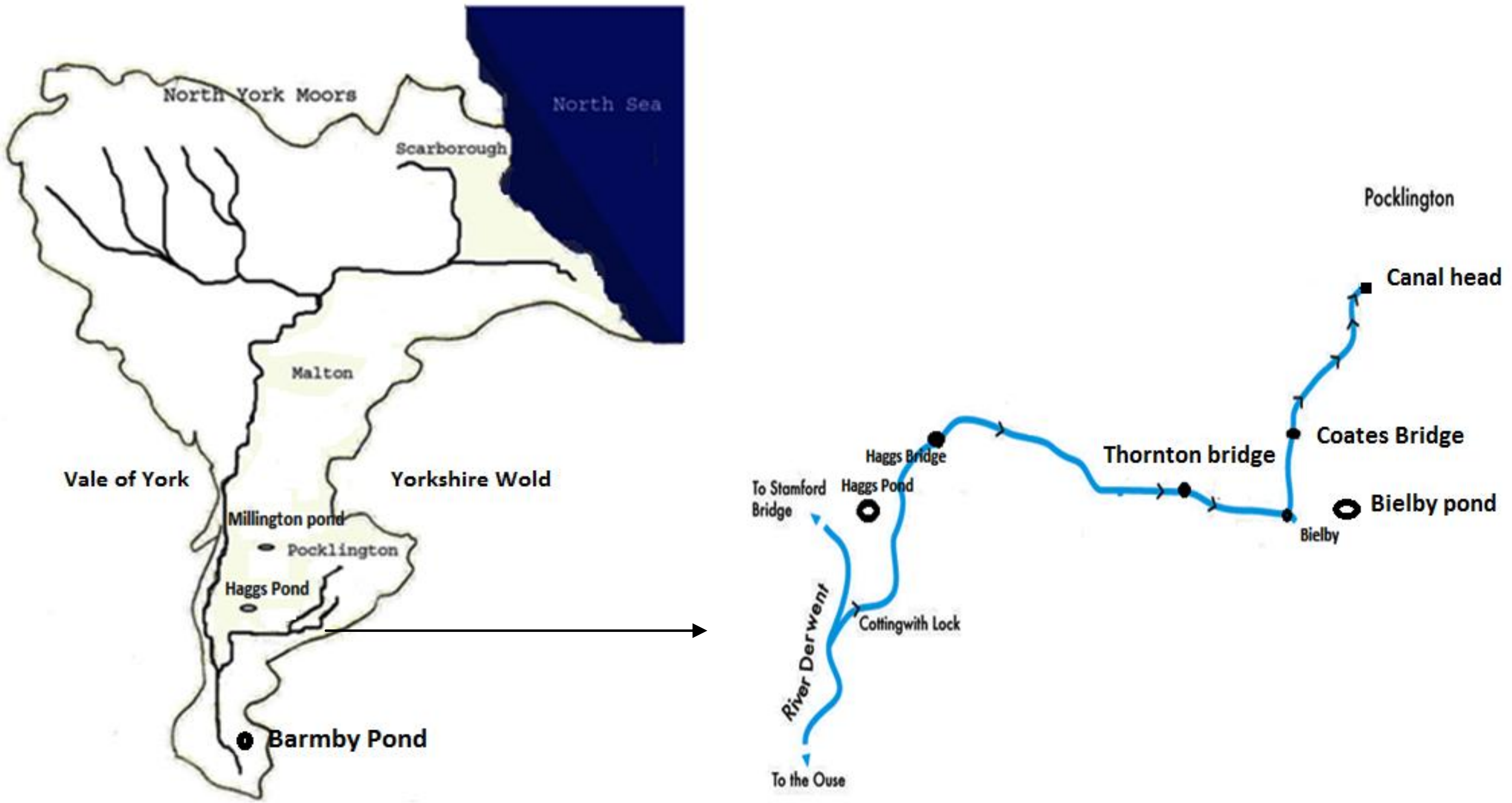


Figure 5.1. Static sites sampled, which includes canals and ponds. Right: Six sites were sampled along the Pocklington Canal, from Canal head to Haggs Bridge. Left: Barmby, Haggs and Millington Ponds .

5.2 Sampling Sites

Eight static habitats (ponds and canals) were sampled in the autumn (early October 2009): Millington Pond, Coates Bridge (canal), Canal Head, Thornton Church Bridge (canal), Haggs Bridge (canal), Bielby Pond, Haggs Pond and Barmby Pond.

5.2.1 Millington Pond

Millington is a small and attractive village in the East Yorkshire Wolds. Millington wood is protected for nature conservation and public access. To left side of Millington wood is Millington pasture, a popular walking area on one of the highest parts of the Wolds. The management for this site is to re-establish natural woodland to allow increases in the ground flora and wildlife of the area. Benthic samples were collected from a small pond located at the base of this area at Millington, which provides drinking water for grazing cattle but is otherwise un-impacted. The water depth here was about 1m.

Figure 5.2. MillingtonPond

Photo: Dave Raffaelli

5.2.2 Canal Head

This site is the terminus of the Pocklington canal which runs from Canal head to the River Derwent, is heavily silted and intruded by weeds and reeds. The area attracts anglers mainly for tench and roach which reach a good size (Anonymous, 2010). The water depth here is about 1m.

Figure 5.3. Pocklington at Canal Head

Photo: Dave Raffaelli

5.2.3 Coates Bridge

Coates Bridge supports a minor road over the Pocklington canal. The samples were collected from the canal passing under this bridge where the bed is a mixture of clay and gravel and water depth is about 2m.

Figure 5.4. Coates Bridge

Photo: Dave Raffaelli

5.2.4 Thornton Church Bridge

Thornton village is located about 16 km southeast of York. The area is surrounded by green fields and farmland. Samples were collected from the canal which passes under Church Bridge, and the bed is mostly clay and stony. Water depth here is 2m.

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Figure 5.5. Thornton Church Bridge

Photo: Dave Raffaellli

5.2.5 Haggs Bridge

Haggs Bridge is a main road bridge and crosses the Pocklington canal about a mile outside Sutton-on-Derwent. The canal here also provides a good fishery and recreation for anglers as well as a public footpath. The water here is about 2m deep with a muddy bed.

Figure 5.6. Haggs Bridge

Photo: Dave Raffaelli

5.2.6 Haggs Pond

The pond is located adjacent to Haggs Bridge but is isolated from the canal and is surrounded by reeds. It is in the middle of cattle pasture and is used by cattle for drinking. The pond bed is muddy and < 1m deep at the time of sampling.

Figure 5.7. Haggs Pond

Photo: Dave Raffaelli

5.2.7 Bielby Pond

Bielby is a small village in East Yorkshire situated about 6.4km south of Pocklington. Most houses are residential, and the area is rural farmland. Bielby pond is on private farmland and is a man-made pond used for storing drinking water for cattle and also a refuge for ducks for occasional shooting. It has a muddy/clay bed with little vegetation. The water depth at the time of sampling was 0.5m.

Figure 5.8. Bielby Pond

Photo: Dave Raffaelli

5.2.8 Barmby Pond

Barmby is a small village in North Yorkshire located 2.5 km west of Pocklington. The area is bordered with open space and countryside with farmland. Barmby pond is quite large with woodlands and reeds surrounding the edge. Water depth at the time of sampling was 0.5m.

Figure. 5.9. Barmby Pond

Photo: Dave Raffaelli

5.3 Methods

Surber and kick sampler are not appropriate for sample collection in static water bodies, so a pond net fitted with a 200µm mesh (described in chapter 2) was used. In this method the net was pushed for 5 metres through the top few cm of substrate (mud) and the large area covered and volume of material collected in this way was such that only one large sample (representative) was collected at each of the sites. Samples were brought to the laboratory and preserved in 70% ethanol. The animals were separated from debris using 250µm mesh sieve and identified to the lowest taxonomic level (detailed in Chapter 2).

Biological indices, such as Community Abundance, Biological Monitoring Working Party Score (BMWPS) and Average Score per Taxon (ASPT), were determined to evaluate water quality. It should be noted that these biological indices were never designed to determine the ecological conditions in static water habitats and their interpretation must be made with caution. Water samples for each site were also collected in 50ml plastic bottles, labelled, brought back to the laboratory and refrigerated for future analysis, specifically conductivity, known to be an important measure of water quality for ponds. However, unlike biological indices, conductivity only reflects one aspect of water quality at the specific time of sampling.

5.3.1 Conductivity measurement

Conductivity is a measurement of the ability of an aqueous solution to carry an electric current. Pure water is not a good conductor of electricity, and conductivity increases as the concentration of ions increases in the solution. A linear relationship has been found between electrical conductivity and level of pollution in lake water (Das *et al.*, 2006). Conductivity was determined using a conductivity meter (H_1 9033, Multi-range conductivity meter, HANNA Instrument) which measures turbidity and dissolved salts in µS/cm. Typical conductivities of different water conditions are given below.

Solution	**µS/cm**
Totally pure water	0.055
Typical DI (deionised) water	0.1
Distilled water	0.5
Domestic “tap” water	500-800
Potable water (max)	1055
Brackish water	100,000

The conductivity meter was first calibrated by immersing into two type of standard buffer solution (tetraborate pH= 10, phosphate, pH= 7) which most closely matched the expected conductivity of the water samples to be measured. The minimum depth of the solution was 8 cm. The readout was then adjusted to match the value given at 25˚C using the trimmer screw on top of the meter. The conductivity standards were 1413µs/cm at 25˚C and all subsequent readings were compensated to this temperature. The bottom of the electrode was tapped on the beaker when any reading was taking to remove any air bubbles and ensure the sample covered the holes on the PVC cover of the probe. Five minutes were allowed for equilibrium of the sample. The lowest range in the meter was selected and the probe was inserted into the samples to above the hole, stirred and tapped until a stable reading was obtained. If a value of ‘1’ was displayed on the left hand side of the display, the next range was tried upon the meter. The procedure was continued until a satisfactory result was obtained. The probe was rinsed with deionised water between readings to reduce chances of contamination.

5.3.2 Body Mass Determination

Body sizes of invertebrates were determined under a low power microscope fitted with a graticule eyepiece. Body dimensions (body length, head width, body width) were measured and dry mass was calculated using allometric equations (Chapter 2), approximate 2-D or 3-D geometric shapes and by water displacement. Size distributions were analysed by Kernel Density Estimation (KDE) and the Body Mass Difference Index (BMDI), as detailed in chapter 2.

5.4 Results

5.4.1 Taxon Richness

A total of 31 families of invertebrates were identified at these sites (Figure 5.10, Table 5.1). The highest number of taxa (families) was found at Millington Pond and Church Bridge and the least at Haggs Pond. Similar numbers of taxa were found in Coates and Haggs Bridge, numbers decreasing from Canal head to Barmby Pond and then Bielby. The highest number of individuals was present at Canal head and the fewest at Bielby. No single site shows dominance of the indicator taxa EPT (Ephemeroptera, Plecoptera and Trichoptera), but these are more typical of stream (flowing) environments. Millington Pond is dominated by Chironomidae, Gammaridae, Asellidae Hydrobiidae and Beetle Larvae (Haliplidae). At Coates Bridge, there was an abundance of Asellidae and Notonectidae while at Canal head large numbers of Asellidae (1359) were found which made 84 percent of the total. At Thornton Church Bridge there was an abundance of Asellidae, Sphaeriidae, Chironomidae and Notonectidae. Corophidae, Hydrobiidae, Asellidae and Sphaeriidae made a large proportion of the fauna at Haggs Bridge. Although the numbers of individuals was very low at Bielby, the families Asellidae and Polycentropidae were abundant here. At Haggs Pond, 78 percent of individuals were Physidae, while Chironomida made up a large proportion of individuals at Barmby Pond.

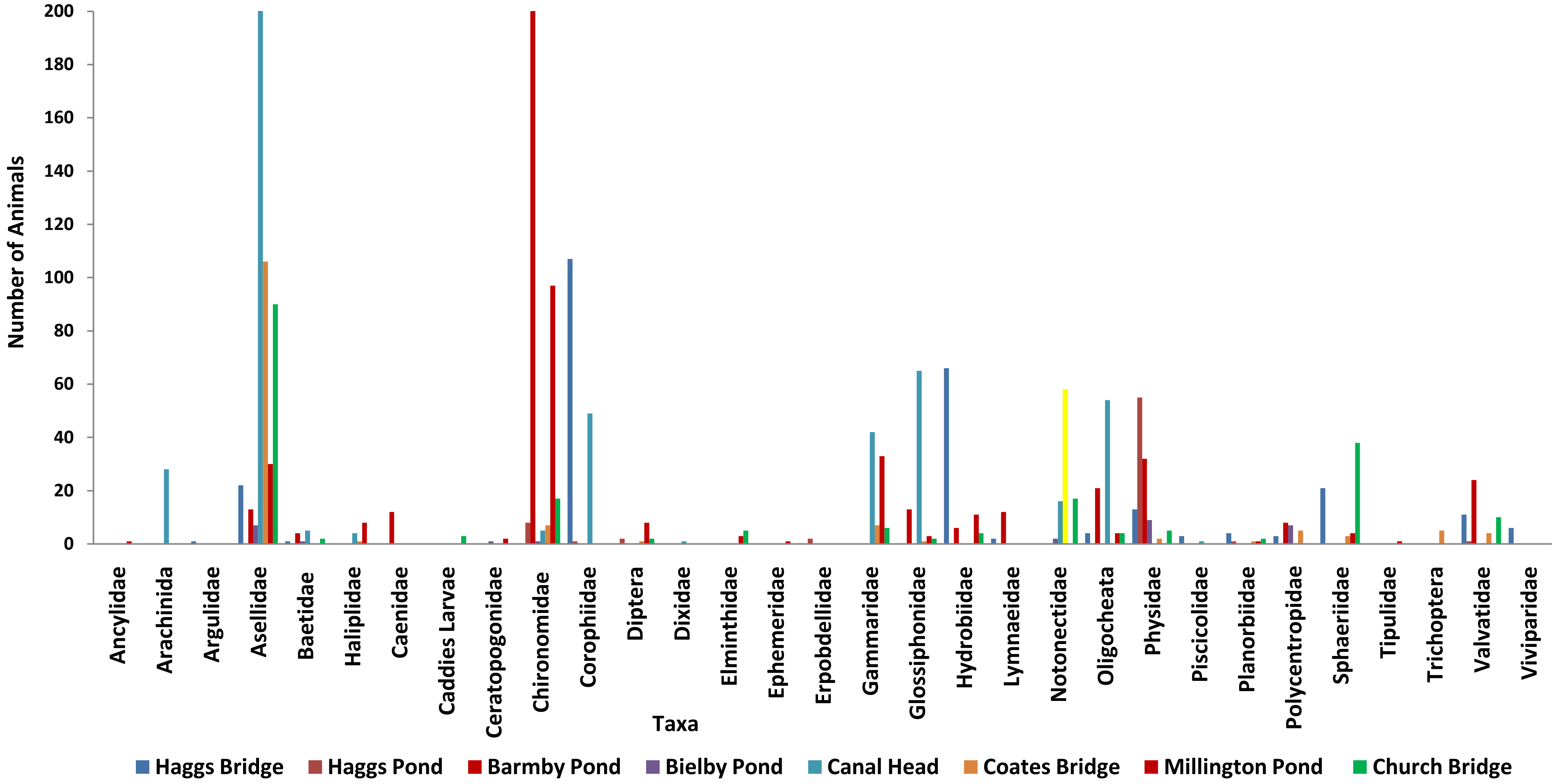

Figure 5.10. Abundance of macroinvertebrates for different sites in the Pocklington canal, Millington Pond, Bielby Pond, Barmby Pond and Haggs Pond.

Table 5.1. The number of individuals recorded at static water sites.

Taxa	Millington	Coates	Canal Head	Church Bridge	Haggs Bridge	Bielby	Haggs Pond	Barmby
Ancylidae	1	0	0 0	0	0	0	0	0
Arachinida	1	0	28	0	0	0	0	0
Argulidae	0	0	0	0	1	0	0	0
Asellidae	30	106	1359	90	22	7	0	13
Baetidae	0	0	5	2	1	1	0	4
Haliplidae	8	1	4	0	0	0	0	0
Caenidae	0	0	0	0	0	0	0	12
Caddies Larvae	0	0	0	3	0	0	0	0
Ceratopogonidae	2	0	0	0	0	1	0	0
Chironomidae	97	7	5	17	0	1	8	217
Corophiidae	0	0	49	0	107	0	1	0
Diptera	8	1	0	2	0	0	2	0
Dixidae	0	0	1	0	0	0	0	0
Elminthidae	3	0	0	5	0	0	0	0
Ephemeridae	1	0	0	0	0	0	0	0
Erpobdellidae	0	0	0	0	0	0	2	0
Gammaridae	33	7	42	6	0	0	0	0
Glossiphonidae	3	1	65	2	0	0	0	13
Hydrobiidae	11	0	0	4	66	0	0	6
Lymnaeidae	0	0	0	0	2	0	0	12
Notonectidae	0	58	16	17	0	2	0	0
Oligocheata	4	0	54	4	4	0	0	21
Physidae	0	2	0	5	13	9	55	32
Piscicolidae	0	0	1	0	3	0	0	0
Planorbiidae	1	1	0	2	4	0	1	0
Polycentropidae	0	5	0	0	3	7	0	8
Sphaeriidae	4	3	0	38	21	0	0	0
Tipulidae	1	0	0	0	0	0	0	0
Trichoptera	0	5	0	0	0	0	0	0
Valvatidae	0	4	0	10	11	0	1	24
Viviparidae	0	0	0	0	6	0	0	0
Total individuals	**207**	**201**	**1629**	**207**	**264**	**28**	**70**	**362**
Total Taxa	**15**	**13**	**12**	**15**	**14**	**8**	**7**	**11**

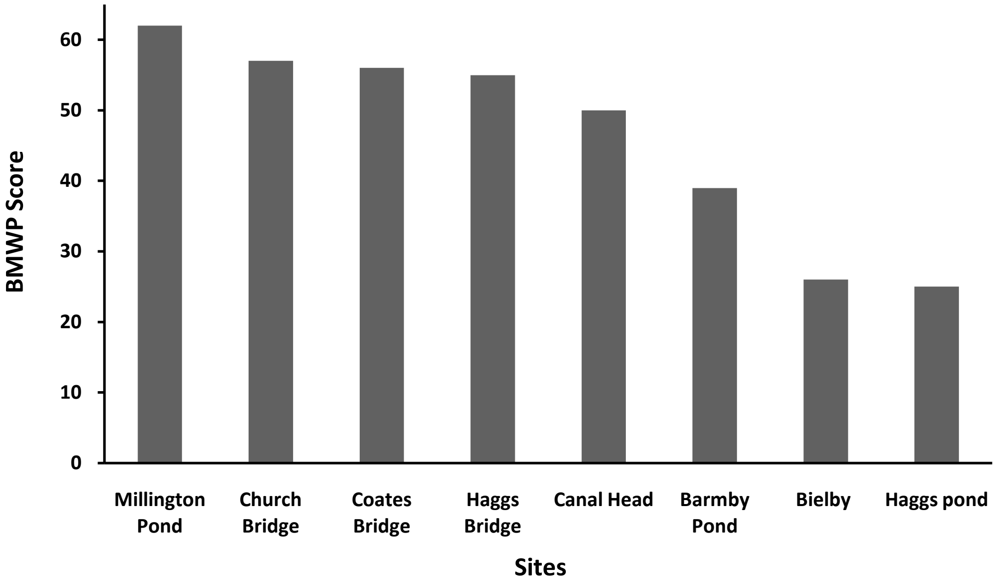

Figure 5.11. Biological Monitoring Working Party (BMWP) scores for ponds and canal sites.

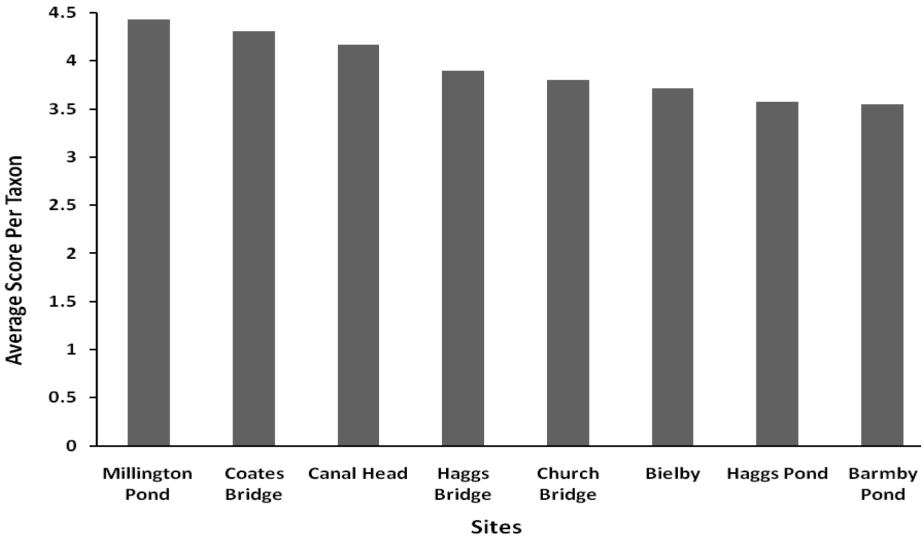

Figure 5.12 Average Score per Taxon (ASPT) for ponds and canal sites.

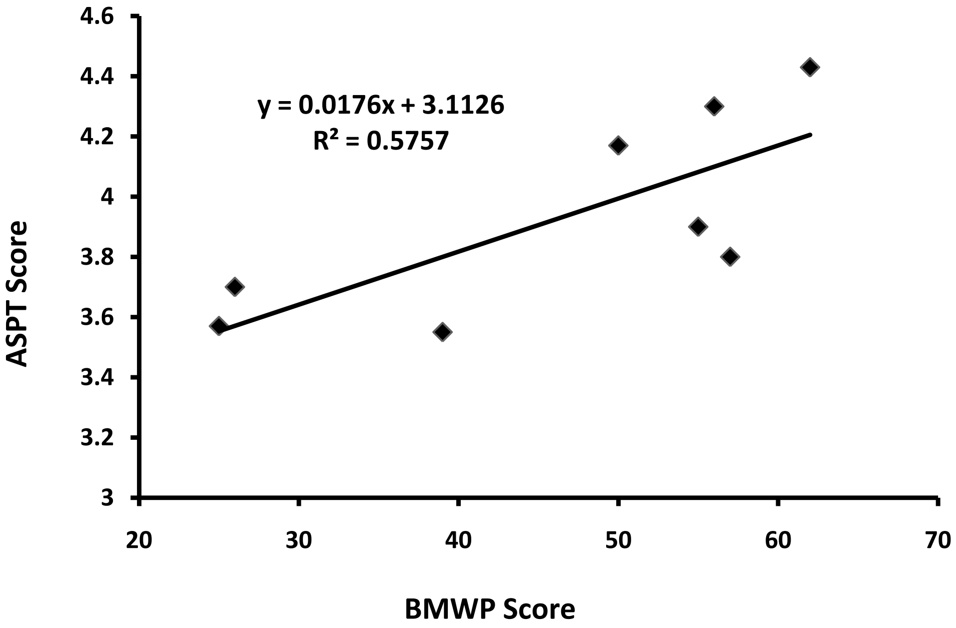

Figure 5.13. The relationship between two biological indices ASPT and BMWP for static sites.

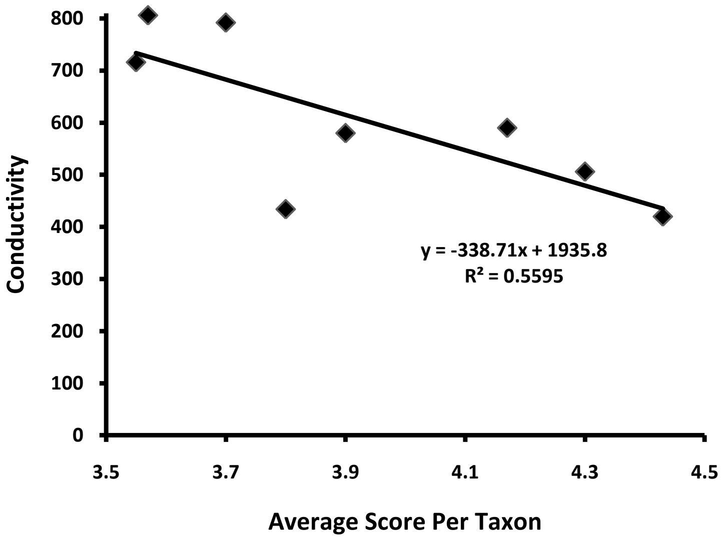

Figure 5.14. The relationship between conductivity and ASPT (water quallity).

5.4.2 Biotic Indices

The Biological Monitoring Working Party (BMWP) score (Figure 5.11) was highest score for Millington Pond (62) with lower values (25) and (26) at Haggs Bridge and Bielby. None of the sites have a value greater than 100, normally associated with clean flowing water systems.

The highest ASPT value was recorded for Millington Pond and the lowest is for Barmby Pond. ASPT values decreases from Coates Bridge to Canal head and from Haggs Bridge to Thornton Church Bridge and the value at Bielby are 3.72 while Haggs pond has almost same value as Barmby Pond. Whilst there are differences in the ranking of sites using ASPT and BMWP, overall the correlation is good between the two techniques (Figure 5.13) $R^2 = 0.5757$, $p < 0.05$.

5.4.3 Conductivity

Conductivity was lowest for Millington Pond while the highest conductivity was recorded for Barmby Pond. These results are similar to the ASPT and BMWP scores for clean and polluted sites but the values for intermediate quality water revealed marked variability with regard ranking of ASPT or BMWP values (Table 5.2). Conductivity values 500- 800 µS/cm are the range for tap water. According to both of the biotic indices and the conductivity data, Millington Pond is of good quality water, while Barmby Pond is of poor quality, but rankings of intermediate quality sites are variable.

Table 5.2. Conductivity measures at 25˚C for water samples from different sites (ponds and canals).

Sites	**Conductivity**
Millington Pond	420µS/cm
Bielby pond	434µS/cm
Haggs Pond	506µS/cm
Coates Bridge	580µS/cm
Canal head	590µS/cm
Haggs Bridge	716µS/cm
Thornton Church Bridge	792µS/cm
Barmby Pond	806µS/cm

5.4.4 Body Mass Patterns

Visual observations of the body mass histograms for most sites show right skewed distributions except for Coates Bridge which shows multimodality in its spectrum (Figure 5.15). The body mass distributions for most sites are dominated by smaller body masses (0.25–0.5mg) and another mode is apparent for large size classes at some sites such as Millington Pond, Canal head and Barmby Pond. There is no indication of such modes for Haggs Bridge, Canal head and Bielby although a minor mode for body mass larger than 5 mg may be present at Thornton Church Bridge.

Kernel density analysis of body mass spectra showed marked differences in modalities between sites. Millington Pond, which has good water quality revealed only one mode, while the most polluted site Barmby Pond had 4 modes (Table 5.3). The intermediate quality sites are highly variable in modality and there is no consistency between the number of modes and ASPT or BMWP score. If anything, the relationship is negative, the reverse of that seen in chapter 3 and 4 for streams. Thus an inverse relationship has been revealed between water quality (BMWP score and ASPT) and the number of modes (Figure 5.17) and the correlation was significant $R^2 = 0.5072$, $p < 0.05$).

Gap analysis using BMDI the approach revealed differences in the number of gaps in clean and polluted sites. The highest number of gaps was found for intermediate followed by the poor quality site. There was no evidence of the effect of water quality on the number of gaps. Thus my results provide no support for a relationship between water quality and body size distributions for static quality habitats (Figure 5.19).

As in Chapter 4, it is important to explore possible phylogenetic effects on the creation and location of modes in the body mass distributions observed static water bodies. The taxa associated with modes are shown for very clean, intermediate and very polluted sites in Figure 5.20. Millington Pond, a clean site had only one peak, which is a mixture of six taxa. Similarly, in intermediate quality site (Haggs Bridge) peak 1 is entirely dominated by the Hydrobiidae but peak 2 and trough 1 contains 5 or 6 taxa in the two body mass modes. In the very polluted site (Haggs Pond), two peaks (Peak 1 and Peak 3) dominated by Physidae but peak 2 and trough 2 contain a mixture of 3 or 4 taxa.

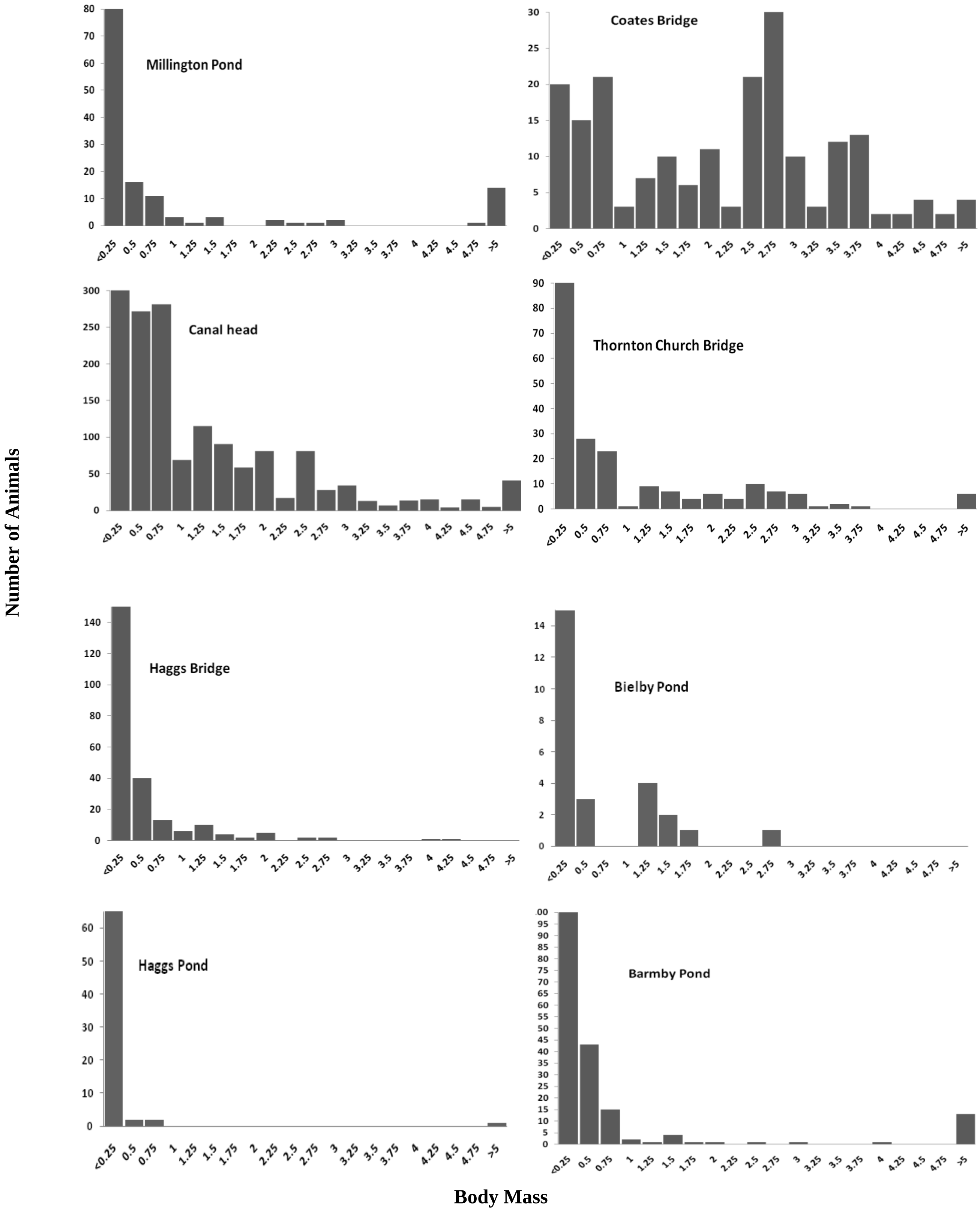

Figure 5.15. Frequency histograms of body mass distributions for benthic communities of static sites (pond and canals)

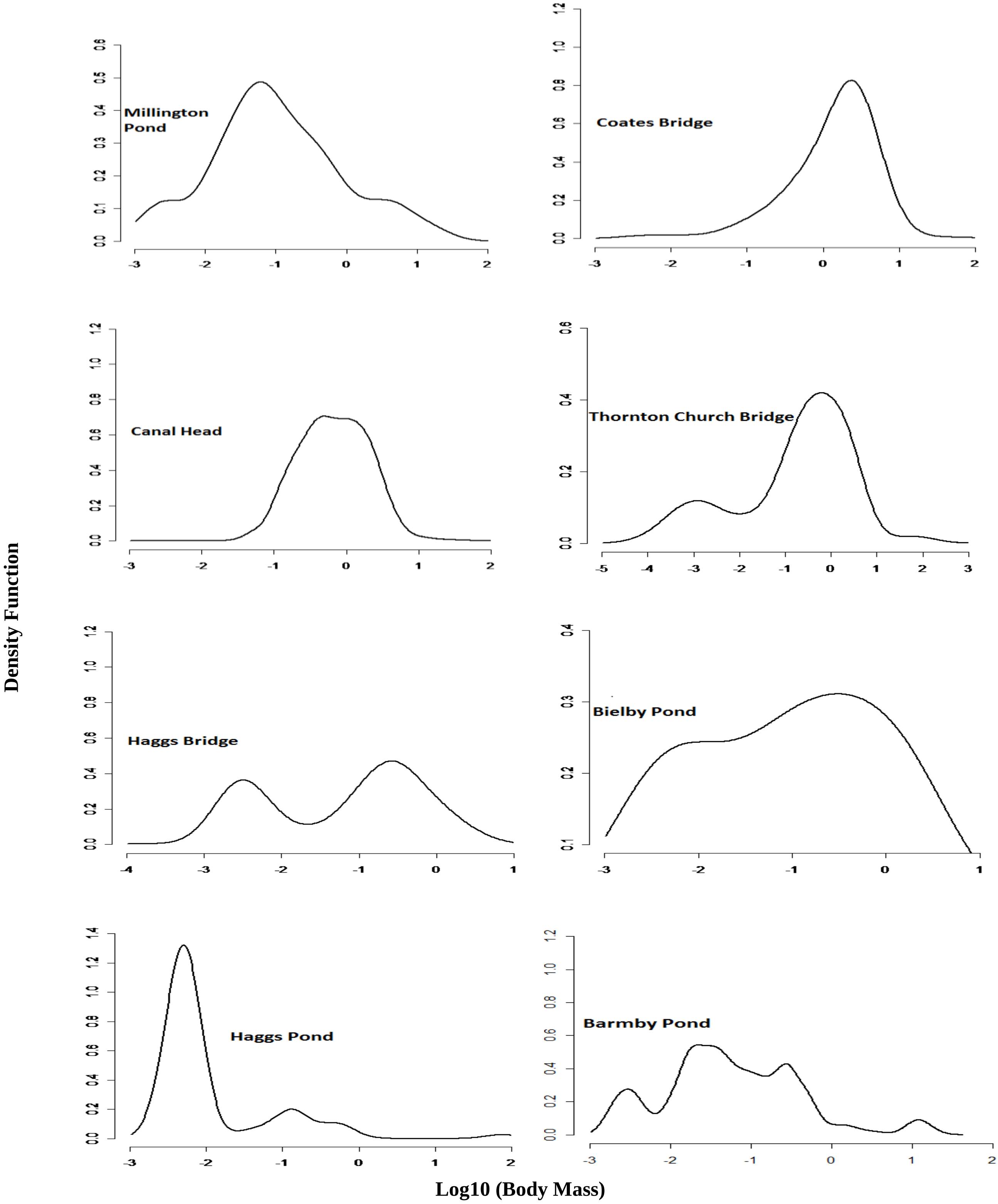

Figure.5.16. The fitted density distributions for body sizes derived by kernel density estimation and bootstrapped re-sampling for the static sites (ponds and different locations on the Pocklington canal). Axes are scaled as $\log_{10}$ of original data. Density function = number of individuals at the site.

Table 5.3. Significance test for body mass spectra from kernel density estimation and smoothed bootstrap re-sampling. The locations of peaks and trough, h = smoothing constant used in kernel estimation, m the smallest number of modes for which the bootstrap test was not significant at 5% level, α level of significance for each distribution (mode number).

Sites	**h**	**m**	**α**
Mllington Pond	0.284	1	0.231
Coates Bridge	0.316	1	0.236
Canal Head	0.327	2	0.052
Thornton Church Bridge	0.386	2	0.075
Haggs Bridge	0.109	2	0.67
Bielby Pond	0.59	1	0.107
Haggs Pond	0.219	3	0.184
Barmby Pond	0.131	4	0.501

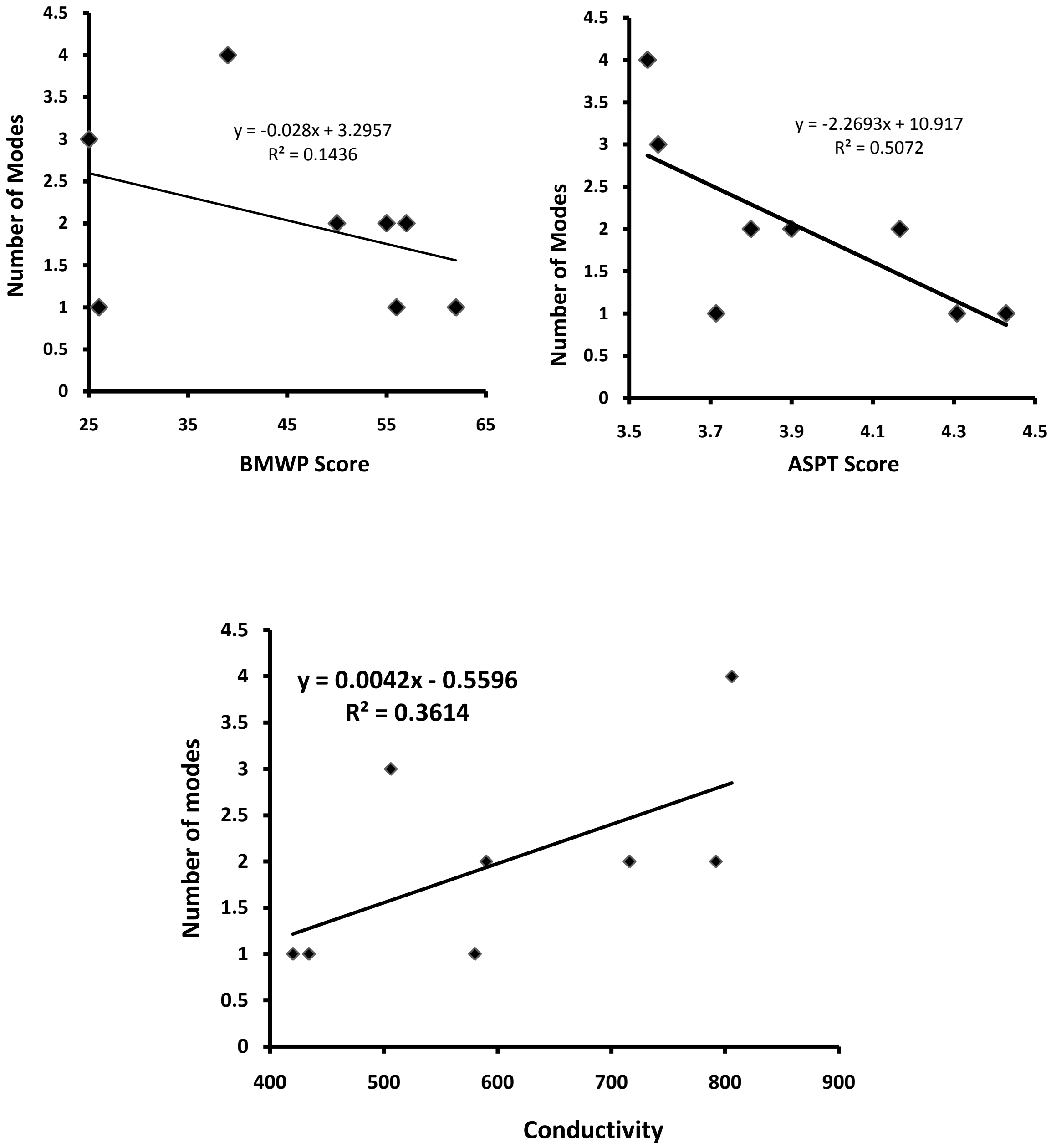

Figure 5.17. The relationships between biotic indices (BMWP and ASPT) and conductivity as water quality and number of modes (determined by Kernel Density Estimation)

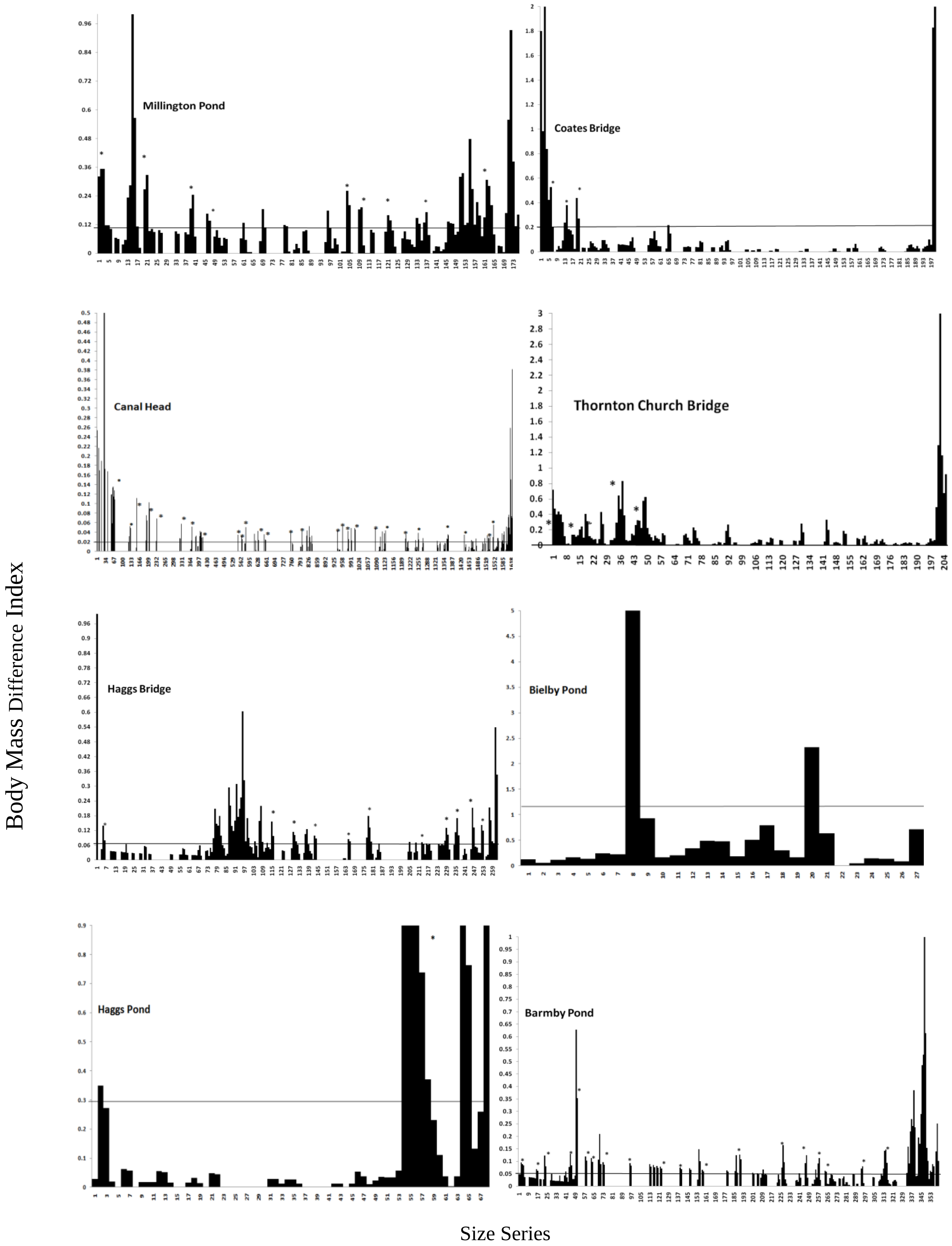

Figure 5.18. Distributions of body mass gaps for static water habitats (Ponds and canals) at Yorkshire. The horizontal line is mean + 2SE and asterisk (*) significant body mass gaps, identified as at least two value above followed four value below the line.

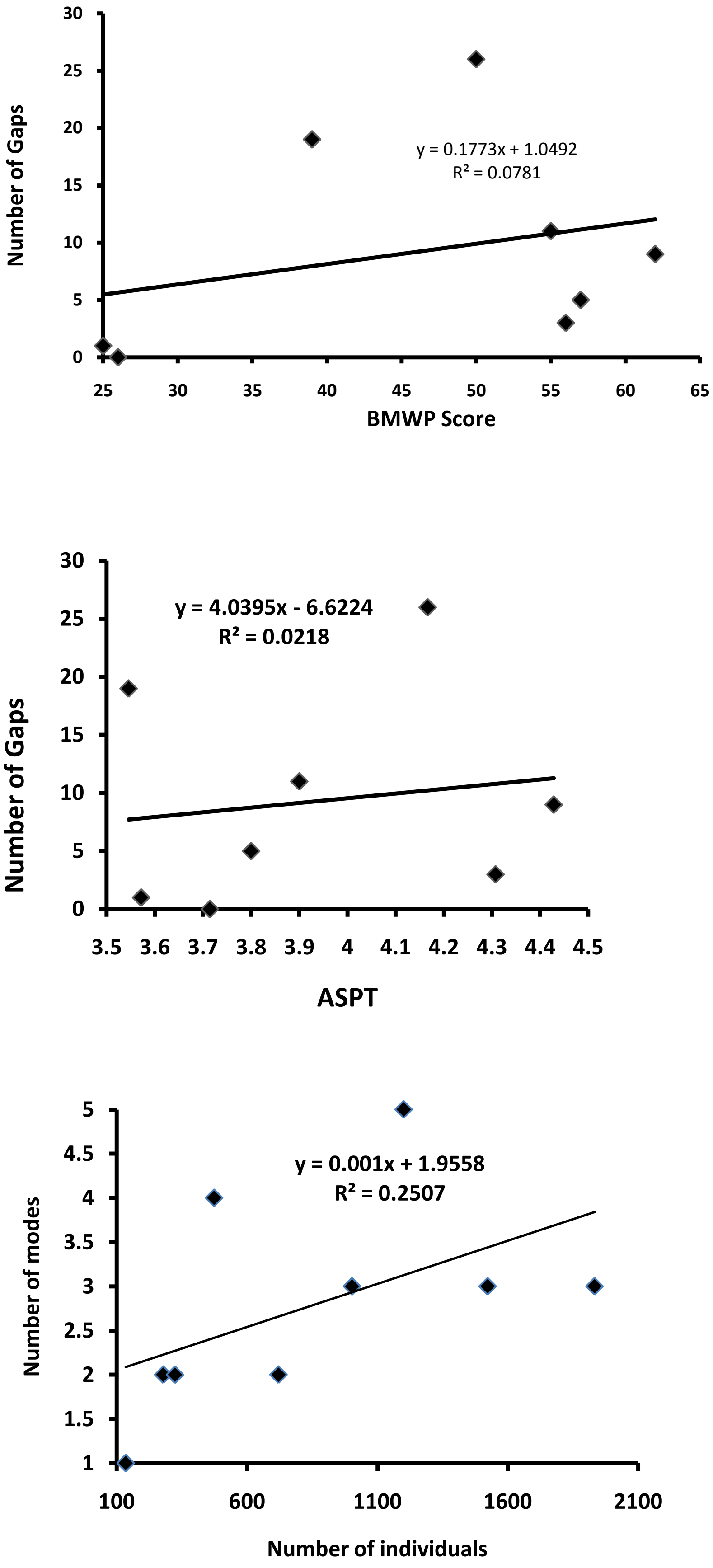

Figure 5.19 The relationship between water quality (BMWP and ASPT) and the number of gaps and the relationship between taxa present and number of modes

Table 5.4. The location of troughs and the number of gaps in body mass distributions determined using two different approaches KDE and BMDI. The locations of gaps are not included due to increased number.

Sites	**Peaks**	**Troughs**	**No of gaps**
Millington Pond	0.061	-	9
Coates Bridge	2.35	-	3
Canal Head	0.493 0.904	0.841	27
Thornton Church Bridge	0.00119 0.601	0.00973	5
Haggs Bridge	0.00391 0.265	0.0216	11
Bielby Pond	0.306	-	-
Haggs Pond	0.00502 0.131 0.42	0.0253 0.391	1
Barmby Pond	0.00291 0.0216 0.266 12.1	0.00638 0.143 5.05	19

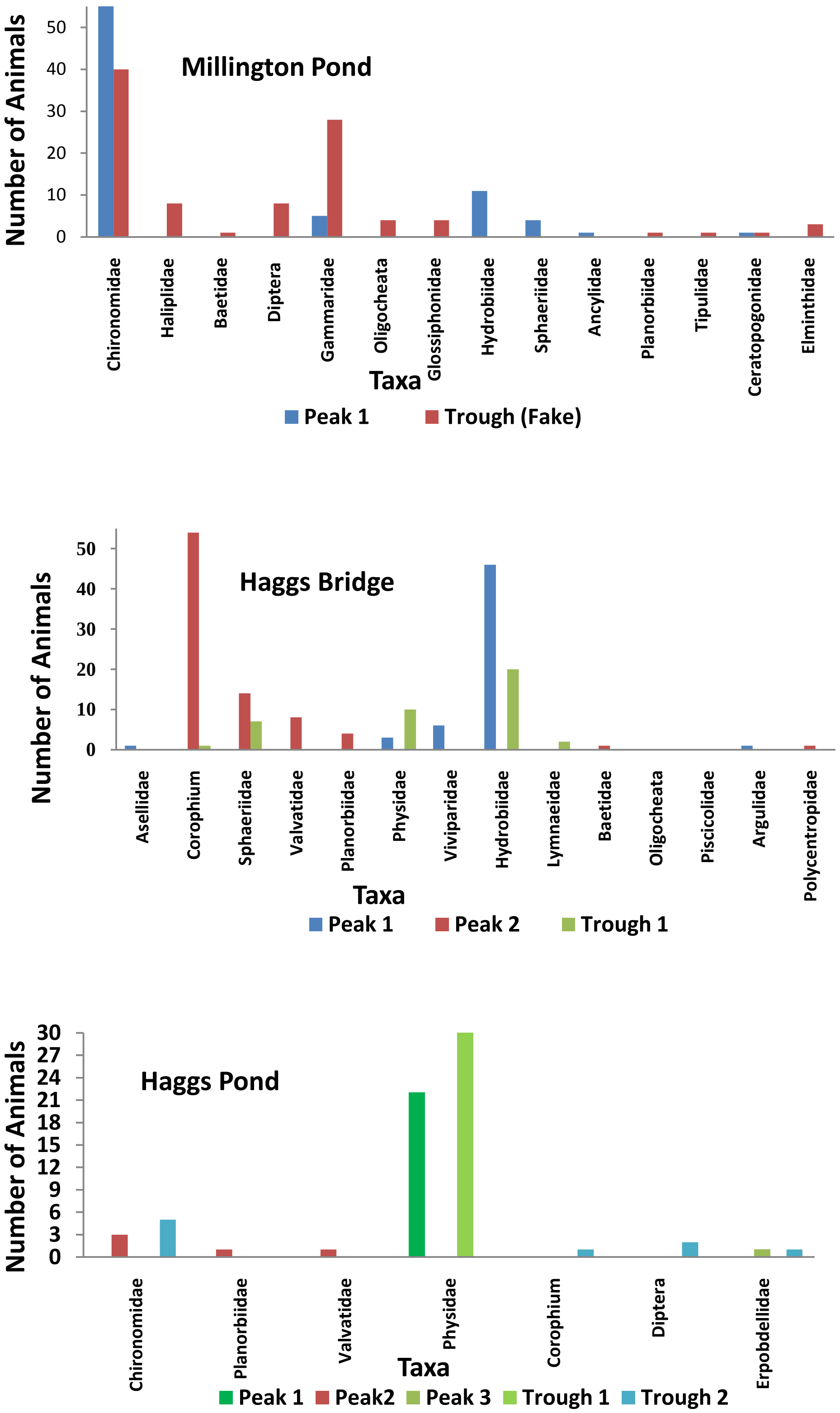

Figure 5.20 The taxonomic groups occurring in body size modes presented with different colours for modes and troughs in static water bodies.

5.5 Discussion
Water quality across the sites

5.5.1 Biotic indices

This analysis suggests a gradient of water quality with Millington Pond the cleanest and Barmby Pond the poorest quality site. The highest number of taxa was present at Millington Pond and Thronton Church Bridge, while the number of individuals are highest at Canal head, dominated by Asellidae. There are far fewer taxa (7) recorded at Haggs Bridge and the lowest number individuals (28) were present at Bielby Pond followed by Haggs Pond which had only 70 individuals. Indicator taxa for clean quality were not dominant at any site and stoneflies were, of course, absent in all samples. However pollution tolerant taxa are significantly higher at Barmby Pond, indicating poor water quality.

The general quality assessment of static water bodies (lakes, ponds, canals and ditches) is difficult due to an absence of standarized assessment methodology, because there has been little monitoring of these water bodies by regulatory agencies like the Environment Agency. However, a combination of water quality assessment approaches is potentially aplicable for static water bodies (Howard, 2002). Taxon richness or rarity can be used to identify the deterioration of a waterbody because communities have strong monometric relationships with degradation (De'ath and Fabricius, 2010). Biotic index (BMWP) for the sites show a higher value 62 for Millington Pond and lower value 25 for Haggs Pond. The value decreases from 57 (Thornton Church Bridge) to 26 for Bielby Pond (Figure 5.11). The widely used biotic index ASPT score for Millington Pond is also the highest which agrees with the BMWP score. ASPT values decrease from Coates Bridge to Canal head and Haggs Bridge, and the lowest score being recorded for Barmby Pond. There is, however, no consistency in BMWP and ASPT values for intermediate sites. This might be due to differences in the number of taxa present at the sites reflecting the qualitative nature of the sampling and different sample sizes, which is a major drawback with the BMWP approach (Gray, 1999).

5.5.2 Water chemistry

Analysis of water chemistry is useful for determining water quality at the time of sampling (Egan, 1976). Conductivity at Millington Pond was 420µS/cm or 0.42mS/cm, which corresponds to values commonly found in natural water. A considerable increase of suspended solids were observed in the Thornton Church Bridge and Barmby Pond samples which had values of 792µS/cm and 806µS/cm, respectively, indicating increased levels of dissolved minerals. The values for intermediate sites are variable but in the lower range (432µS/cm-716µS/cm), indicating a lower concentration of dissolved materals. The Biological index results and the chemical analysis only contradict each other when contamination is intermittent (Whitton, 1991) because, for example, Dissolved Oxygen (DO) is influenced by several parameters including suspended and dissolved solids (Vicente *et al.*, 2009). High correlations have been found between chemical and biological analysis in determining the quality of polluted rivers (Cain *et al.*, 1979). In this study, such correlation found statistically significant (figure 5.14. $R^2 = 0.5595$ $p<0.05$).

5.5.3 Body size distributions

The body size distributions are right skewed for most sites, dominated by smaller body masses in the range 0.25-0.75mg, except for Coates Bridge which shows multimodality in the histograms. There are minor modes for large size class in some sites but not in Haggs Bridge, Bielby and Haggs Pond (Figure 5.15). However visual inspections of body size plots are unreliable, due to possible bin size effects as discussed in the previous chapter.

5.5.4 KDE anlysis

Statistically rigorous estimates of the number of modes revealed marked differences between clean and polluted sites, but in contrast to moving water, the body size spectra at static sites were unimodal for the cleanest site, Millington Pond, while the polluted site, Barmby Pond, had 4 modes (Table 5.3). The number of modes ranges from 1-4 for different static water bodies to give a statistically significant inverse relationship between water quality (ASPT) and number of modes. The present study offer no explanation for this inverse trend. It seems unlikely that the sites have been mis-classified (Millington the dirtiest and Barmby the cleaniest) because both biotic indices and the conductivity measurement do not support that.

5.5.5 BMDI analysis

The number of gaps is 9 at Millington Pond while Barmby Pond, which is considered the most polluted site, has 19 gaps. The highest number of gaps was recorded at Canal head while there is only one gap at Haggs Pond and no gap detected at Bielby Pond (Figure 5.18). Thus BMDI results in this study shows no clear trends, if anything, the opposite of that found in streams in chapter 3 and 4. Most samples were collected from adjacent sites in and around Pocklington canals so there was less restriction for movement between the sites. Thus, the differences in body mass distributions are unlikely to be due to geographical boundaries. The body masses of animals were measured at the family level and these organisms have similar evolutionary histories, so that both the phylogenetic and community interaction hypothesis are unlikely to be responsible for the observed patterns in modality.

5.5.6 Locations of troughs and gaps

Differences have been seen for the number and the locations of gaps and troughs in the body mass distributions (Table 5.4). There was no trough found at Millington pond using the KDE analysis while 9 gaps were detected in BMDI analysis. Similarly at Haggs Pond three troughs and only one gap were recorded. Bielby Pond had presented no troughs and no gaps in the distributions. Thus, there was no consistancy between these two approaches and the relationships of these techniques with the quality of lentic ecosystems is not significant (Figure 5.19).

The size spectra for static water habitats have been reported as strikingly different from those in marine benthic communities (Strayer, 1986), but here they can be seen to be highly variable in the number of modes. Figure 5.20 shows that the number of modes increased with declining water quality. In the cleanest (Millington Pond) and intermediate quality (Haggs Bridge) sites, taxa were scattered across peaks and troughs, but in the most polluted site the taxa are more confined to individual peaks and troughs. Howeverl, there is little evidence that the occurrence of modes and troughs are due to phylogenetic effects but the over all relationship between the number of taxa and and number of modes were insignficant (Fig 5.19)

5.6 Conclusion

Static water bodies show variation in the number of modes and the number of gaps in body size spectra of benthic communities, as found in streams. However, but this does not appear to be related to water quality. In fact, any relationships seem to be the reverse of those found for stream (flowing) habitats in chapter 3. The significance of this is discussed further in chapter 6.

5.7 References

Anonymous (2010).York and district amalgamation of anglers.

Boisclair, D. and Leggett, W. C. (1985). Rates of food exploitation by littoral fishes in a mesotrophic north-temperate lake. *Canadian Journal of Fisheries and Aquatic Sciences* 42: 556-566.

Cain, J. R., Lawrence Klotz, R., Trainor, F. R. and Costello, R. (1979). Algal assay and chemical analysis: A comparative study of water quality assessment techniques in a polluted river. *Environmental Pollution (1970)* 19(3): 215-224.

Chatelliers, M. C. D. and Reygrobellet, J. L. (1990). Interactions between geomorphological processes, benthic and hyporheic communities: First results on a by-passed canal of the french upper rhône river. *Regulated Rivers: Research & Management* 5(2): 139-158.

Cosser, P. R. (1989). Water quality, sediments and the macroinvertebrate community of residential canal estates in south-east queensland, australia: A multivariate analysis. *Water Research* 23(9): 1087-1097.

Das, R., Samal, N. R., Roy, P. K. and Mitra, D. (2006). Role of electrical conductivity as an indicator of pollution in shallow lakes. *Asian Journal of Water, Environment and Pollution* 3(1): 143-146.

De'ath, G. and Fabricius, K. (2010). Water quality as a regional driver of coral biodiversity and macroalgae on the great barrier reef. *Ecological Applications* 20(3): 840-850.

Egan, H. (1976). Chemical analysis and water quality. *Fresenius' Journal of Analytical Chemistry* 282(4): 323-329.

Gray, N. F. (1999). *Water quality assessment in: Water technology: An introduction for environmental scientists and engineers* London: Elaine Wigzell.

Hall, D. J., Cooper, W. E. and Werner, E. E. (1970). An experimental approach to the production dynamics and structure of freshwater animal communities. *Limnology and Oceanography* 15(6): 839-928.

Hanson, J. M. (1990). Macroinvertebrate size-distributions of two contrasting freshwater macrophyte communities. *Freshwater Biology* 24(3): 481-491.

Hanson, J. M., Prepas, E. E. and Mackay, W. C. (1989). Size distribution of macro-invertebrates communities in fresh water lake

Canadian Journal of fishries and aquatic science 46: 1510-1519.

Howard, S. (2002).A guide to monitoring the ecological quality of ponds and canals using psym. Pond Action (Pond Conservation Trust: Policy and Research) Environment Agency Midland Region.

James, M. R., Weatherhead, M., Stanger, C. and Graynoth, E. (1998). Macroinvertebrate distribution in the littoral zone of lake coleridge, south island, new zealand-effects of habitat stability, wind exposure, and macrophytes. *New Zealand Journal of Marine and Freshwater Research* 32(2): 287 - 305.

McAbendroth, L., Ramsay, P. M., Foggo, A., Rundle, S. D. and Bilton, D. T. (2005). Does macrophyte fractal complexity drive invertebrate diversity, biomass and body size distributions? *Oikos* 111(2): 279-290.

Morin, A. and Nadon, D. (1991). Size distribution of epilithic lotic invertebrates and implications for community metabolism. *Journal of the North American Benthological Society* 10(3): 300-308.

Rasmussen, J. (1993). Patterns in the size structure of littoral zone macroinvertebrates communites *Can J Fish Aquat Sci* 50: 2192-2207.

Rath, L. C. (1986).Le rapport entre heterogenite due benthos et les sediments lacustres dans les zones profondes et littoriprofodes. In *University de Montreal*, Vol. MSC thesis.

Strayer, D. (1986). The size structure of a lacustrine zoobenthic community. *Oecologia* 69(4): 513-516.

Strayer, D. L. (1991). Perspectives on the size structure of lacustrine zoobenthos, its causes, and its consequences. *Journal of the North American Benthological Society* 10(2): 210-221.

Vamosi, J. C. and Vamosi, S. M. (2007). Body size, rarity, and phylogenetic community structure: Insights from diving beetle assemblages of alberta. *Diversity and Distributions* 13(1): 1-10.

Vicente, J., Colmenarejo, M. F., Sánchez, E., Rubio, A., García, M. G., Borja, R. and Jimenez, A. M. (2009). Evaluation of the water quality in the guadarrama river at the section of las rozas-madrid, spain. *Water and Environment Journal*.

Whitton, B. A. (1991). *Aims of monitoring. In: Whitton, b.A., rott, e., friedrich, g. (eds.), use of algae for monitoring rivers.*: Institut fur Botanik, Universitut Insbruck, Austria.

Chapter 6

General Discussion

6.1 Ecological implications of body size

Animal body size is probably the single most important attribute in the ecological literature. At the community level of organization, body size is an important factor in determining and controlling the course of life, especially for aquatic animals. The size based approach provides a tool for community analysis and for taxonomic techniques, presenting an alternative prospective (Rasmussen, 1993). Numerous studies have shown strong relationships between biological processes (e.g. metabolism, biomass production, home range size, abundance, nutrient recycling, etc) and an individual's body size (Peters, 1983b; Brown *et al.*, 2004), and these are often expressed in the form of a power function

$$Y = aM^b$$

Where Y is the predicted biological character of an individual, M is the body mass of the animal, and a and b are the normalization constant and scaling exponent, respectively (Marquet *et al.*, 2005). Peters (1983) found most biological processes are an allometric function of body size. These relationships can also be expressed in logarithmic form as

$$\log Y = \log a + b \log M$$

Where constant log a, corresponds to the Y-intercept and the value of b is the slope of this line. In many studies of body mass and physiological processes the value of b is ¾ of the whole organism rate and -¼ for mass specific rates (Peters, 1983b; Scmidt- Nielsen, 1984; Brown *et al.*, 2004). The fact that allometric relationships of body size and processes occur at the population, community and ecosystem level emphasizes the importance of body size at all levels of organization. However, neither empirical nor theoretical structure has emerged that adequately explains this phenomenology at the level of the community and above. The primary tool for the investigation of body size structure at the community level is the construction of size spectra which can be expressed in a variety of formats.

6.2 Construction of size spectra

Size spectra were first used to characterize certain size ranges of marine plankton (Sheldon and Parsons, 1967b), marine benthic communities (Schwinghamer, 1981b) and in freshwater systems (Peters, 1983a). Body sizes in benthic communities are often based on direct measurement of a particular body dimension, volume, dry mass or the equivalent spherical diameter, which is the diameter of sphere that has same volume as the measured organism.

There are several methods for constructing a body size spectrum from these data which relate to the underlying community structure, such as total biomass, abundance, number of individuals and the species size spectrum, and spectra can be presented in a number of ways, such as a Regular Body Size Spectrum, a Relative Size Spectrum, a Normalized Size Spectrum and a Cumulative Size Spectrum. The absolute values in each body size class refer to the regular body size spectrum. Converting these values into percentages allows direct comparison of the shapes of two or more spectra, producing a relative size spectrum. Construction of normalized size spectra has been considered the best method to produce the right width of size classes by dividing the summed total in each size class by its width (Platt and Denman, 1978). The size distributions of all size classes are useful in estimating the relative contributions of all size classes and are expressed as a cumulative size spectrum. The criteria of measurement of individuals depend on the nature of the study. Certain groups within the marine benthic community have been included for size spectra e.g. (Parry *et al.*, 1999) while other groups (e.g. juveniles) have been excluded in others (Warwick, 1984). Some size spectra only include specific taxa (Vanaverbeke *et al.*, 2003), and size spectra can be constructed with a single measurement of body size (usually adult) for each species.

6.3 Size approaches used in the current study

In the current study the sizes of individual organisms, rather than the main size of their taxonomic units, were used to construct body mass spectra for freshwater benthic communities. The size classes were based on body mass (dry weight) and expressed in milligrams (mg). Any individual present in a sample contributes to the functioning and dynamic of that group and no distinctions were made between juveniles and adults. During sampling and processing, some organisms were broken into parts, but only the fractions with a head were measured and allocated into their respective size classes. Although including fragmented body parts can generate biased results, the low number of such fragments makes it unlikely that they have significant influence on the shape of size spectra.

Three approaches to detecting modality in body mass distributions were used: visual inspection of histograms, Kernel Density Estimates (KDE) and Holling‘s (1992) approach, the Body Mass Difference Index (BMDI). The body size spectra observed in numerous studies have generated several hypotheses about the factors controlling the distributions of body mass across ecosystems and their modalities. Thus, Schwinghamer (1981) demonstrated a trimodal body mass distribution separated by biomass troughs which reflect the transition between different major groups of marine intertidal benthos (micro, meio and macro fauna). Further, investigation of size distributions in the coastal sublittoral also produced similar trimodal size spectra (Schwinghamer, 1983; Schwinghamer, 1985) which have also been reported in many other benthic studies (Giere, 1993). Changes in sediment characteristics (sediment pore, space size) were thought to affect size distributions (Schwinghamer, 1985) which provides theoretical support in terms of organism’s perception of their environmental dimensions (Silvert, 1996), as well as confirming discontinuous distributions in body mass patterns (Holling, 1992). Schwinghamer’s hypothesis is now a part of main stream benthic ecology literature, but his claims for trimodality are based entirely on visual inspection of histograms where the x-axis (body size) is a series of logarithmic bins. Other benthic studies have frequently produced contrasting patterns. Investigations into the effect of habitat architecture on biomass and abundance size spectra show no shifting in size distributions with changes in granulometry (Duplisea and Drgas, 1999; Leaper *et al.*, 2001; Strayer, 1986). The construction of body size spectra which are rigorously analysed for their modality can help to elucidate the relationships between size and environmental conditions. In this thesis kernel density estimation and smoothed bootstrap resampling were combined to test for the presence of clumps in body mass distributions and the Body Mass Difference Index (BMDI), as used Holling’s (1992), was used to detect the lumpiness and for exploring his Textural Discontinuity Hypothesis (TDH).

6.4 Summary of the main findings of this thesis

Body mass spectra for the River Ure and Foss systems show much evidence for multimodal distributions as well as association of modality and water quality: there were significant relationships between body mass patterns and water quality, the number of modes defined by KDE decreasing with decreasing water quality (chapter 3). The numbers of gaps detected by BMDI between very good and poor quality water show some association with water quality, but very variable results were obtained for sites of intermediate water quality.

It is hard to test Holling's TDH hypothesis at this smaller scale, but work in marine benthic systems suggests major advantages in this respect (Raffaelli and Moller, 1999). Discontinuities in size spectra seem to related to equal discontinuities in habitat architecture (Schwinghamer, 1981b), consistent with Holling's Textural Discontinuity Hypothesis (TDH). Subsequent investigations of marine biomass and abundance size spectra failed to find any changes in body mass distributions when habitat architecture was manipulated experimentally (Leaper *et al.*, 2001). Other work has shown seasonal shifting in the number of modes and number of gaps, in other words that body mass patterns changed seasonally (Schmid *et al.*, 2002; Gaedke, 1992). My own findings that changing patterns in size distributions and the number of gaps in samples obtained with different sampling methods is consistent with the idea that this problem leads to an incorrect characterization of aquatic systems (Pechman *et al.*, 1991; Kerans *et al.*, 1992).

The relationship between modality, gaps and water quality was further tested for the River Aire. The river has been well documented by the Environment Agency and water quality varies at different stations. Nine stations along the river, ranging from very good to poor quality were sampled. Different biological indices for the sites provided similar information on water quality, confirming a gradient of water quality at these sites. Visual observation of histograms of size distributions indicated unimodality for most of them, as reported in other studies (Strayer, 1986; Ahrens and Peters, 1991). However, statistically more rigorous analysis, using KDE and BMDI, revealed multimodality for the high quality site and fewer numbers of modes and gaps were detected for the most polluted site. The distributions of body mass clearly indicated lumpiness in an undisturbed ecosystem, although the patterns for intermediate quality sites were less consistent.

Body mass distributions were also evaluated for static water bodies (canals and ponds) in chapter 5. Analysis of the number of modes and gaps in these benthic assemblages did not support any association between size spectra and water quality. However, body mass distributions were still discontinuous for most sites. Thus, these analyses showed that body mass distributions of benthic communities are lumpy in ecosystems, but further investigations are needed to explore relationships between size distributions and environmental quality. I can offer no explanation for this findings for static water bodies which is the opposite to that for streams. It is possible (but unlikely) that the water quality of these sites has been misclassified because of ASPT or BMWP were not designated for static water. However the sites were selected on the basis of knowledge of sites by my supervisor (Raffaelli) and Barmby is certainly the poorest and Millington the best water quality.

6.5 Alternative hypotheses to explain body mass patterns

There are number of competing hypotheses to explain body mass patterns in ecosystems. The main hypotheses are (i) energetic; (ii) phylogenetic; (iii) biogeographical; (iv) textural discontinuity; and (v) community interaction hypothesis. These are now discussed in relation to the data obtained in the present study.

6.5.1 Energetic hypothesis

The energetic hypothesis is based on the allocation of energy for species growth and their reproduction processes which is limited by resource availability from the environment. Changes in resource availability is expected to confer multiple modes onto body mass distributions (Marquet *et al.*, 1995). Energetic constraints on body mass distributions have been observed in several studies (Morand and Poulin, 2002; Kozlowski, 1996), other researchers have rejected the model (Perrin, 1998; Symonds, 1999). In the present study, body mass patterns have been described within the same river system at sites with different ecological states, so that the availability of energy will be different because of the different local environmental conditions, not because of differences in the utilization of those resources. Thus, there is no strong evidence for the energetic hypothesis to explain the differences observed in body mass patterns at the different sites in the present study.

6.5.2 Phylogenetic hypothesis

Evolutionary processes have been considered to constrain body mass patterns in several studies (Gardezi and Silva, 1999; Smith *et al.*, 2004). According to this hypothesis, the fauna present at the continental scale have a range of macro-evolutionary histories and this generates multiple modes in body mass distributions. Study on mammals have revealed shifts in body size from smaller to larger throughout the Tertiary in North American fossils (Alroy, 2000). Also most of the world's mammals exhibit phylogenetically independent distributions of body mass (Gardezi and Silva, 1999) which are right skewed and this has also been observed in North American freshwater fish (Jason and Page, 2003). The data analysed here are not for specific taxa, but for benthic communities where the taxonomy is similar and their evolutionary histories are expected to be the same at all sites. Thus, in the present study, phylogenetics is held constant and is unlikely to be responsible for the discontinuous distributions in body mass patterns observed.

6.5.3 Biogeographical hypothesis

This model proposes that multimodality in body mass distributions is due to geographical boundaries which limit species dispersal between regions and cause restricted sets of species to be present within a given community (Hubbell, 1997). Pyron (1999) found a positive relationship between geographical boundaries and body mass patterns in North American species of fish (suckers and sunfishes). Similarly, gaps in body mass distributions in mammal communities have been suggested to be due to different abilities of species to disperse across biomes (Siemann and Brown, 1999). In contrast, Holling (1992) showed discontinuous distributions of body masses in adjacent regions which have no obvious boundaries to dispersal. However, in the present study, body mass distributions were compared between ecosystems that were spatially connected and allowed species to disperse across those systems. The sites only differed in their ecological state so that it is unlikely that geographical factors cause the discontinuous patterns of body mass seen at this scale.

6.5.4 Community interaction hypothesis

This hypothesis is based on the idea that interactions between species in communities for limited resources produces multimodal body mass distributions (Nummi et al., 2000). When resources are unlimited right skewed distributions in body mass have been recorded for North American mammals (Brown *et al.*, 1993). The formation of gaps in the body mass distributions of wading birds was found to be due to resource competition (Oksanen *et al.*, 1979). Agosta and Janzen (2005) observed multiple selective mechanisms, rather than community interactions, operating on moths, leading to multiple modes in body mass distributions. This hypothesis deals with interactions at the species level, but in the present study, animals were identified to the family level. Thus, it is unlikely that the community interaction hypothesis is responsible for the discontinuous body mass patterns observed.

6.5.5 Textural discontinuity hypothesis

This hypothesis proposes that ecosystems are driven by dominant structuring processes which often differ from each other by at least an order of magnitude in scale and which are discontinuously distributed in space and time. Species that function at distinct scales respond differently to the opportunities provided at these scales and this is reflected in the multimodality of their body mass patterns (Holling, 1992). This has been observed in a variety of habitats (Restrepo *et al.*, 1997; Bakker and Kelt, 2000). Gutiérrez and Iribarne (2004) found associations between body masses and habitat for fish species and similar relationships have been shown in a study of nekton Moreton Bay, Queensland, Australia (Pittman *et al.*, 2004). In the present study interpretation of body mass patterns for multimodality and discontinuity (gaps) in moving water indicates the presence of multiple structuring processes at cleaner sites as revealed by multiple modes and gaps in body mass distributions, while fewer gaps and modes were recorded at disturbed sites, providing good evidence for a lack of dominant processes at these sites. The discontinuous body mass structure of cleaner habitats is thought to provide a discontinuous (more resilient) ecosystem structure, and it could be argued that size-based indicators for assessing the ecological state of ecosystems will help to provide a mechanistic understanding of the role of body size in community assembly, as envisioned by Hutchinson and MacArthur (1959). However, this study was unable to formally relate discontinuities in body mass distributions to water quality in static water bodies, and where in fact the relationships appear to be the opposite of those found for moving water.

6.6 Future Work

The analyses carried at here were based on surveys of sites with differing water qualities, in order to explore the idea that “lumpiness”, defined here by the number of gaps or number of modes, is related to disturbance. A different approach to this question would be to experimentally test this hypothesis, either by allowing freshwater communities to assemble in experimental systems maintained with differing water qualities, or by lowering the water quality of natural stream communities by adding a stress, such as organic material or excess fertilizer. If this was done in a dose-dependent way, then it might reduce the variance in the modality seen in the intermediate water quality sites in the present study.

6.7 References

Agosta, S. J. and Janzen, D. H. (2005). Body size distributions of large costa rican dry forest moths and the underlying relationship between plant and pollinator morphology. *Oikos* 108(1): 183-193.

Ahrens, M. A. and Peters, R. H. (1991). Patterns and limitations in limnoplankton size spectra. *Can. J. Fish. aquat. Sci.* 48: 1967-1978.

Ali, N., Ahmed, E., Othman, A. K. and Othman, J. (2009). Prediction of johor river water quality parameters using artificial neural networks. *European Journal of Scientific Research* 28(3): 422-435.

Allen, C., Gunderson, L. and Johnson, A. R. (2005). The use of discontinuities and functional groups to assess relative resilience in complex systems. *Ecosystems* 8(8): 958-966.

Allen, C. R., Forys, E. A. and Holling, C. S. (1999). Body mass patterns predict invasions and extinctions in transforming landscapes. *Ecosystems* 2(2): 114-121.

Allen, C. R., Garmestani, A. S., Havlicek, T. D., Marquet, P. A., Peterson, G. D., Restrepo, C., Stow, C. A. and Weeks, B. E. (2006). Patterns in body mass distributions: Sifting among alternative hypotheses. *Ecology Letters* 9(5): 630-643.

Alroy, J. (2000). New methods for quantifying macroevolutionary patterns and processes. *Paleobiology* 26(4): 707-733.

Anonymous (1997).In *Local environmental agency plan Swale, Ure and Ouse: Consultation report*, 144.

Anonymous (2009).York rivers.

Anonymous (2010).York and district amalgamation of anglers.

Armitage, P. D., Gunn, R. J. M., Furse, M. T., Wright, J. F. and Moss, D. (1987). The use of prediction to assess macroinvertebrate response to river regulation. *Hydrobiologia* 144(1): 25-32.

Armitage, P. D., Moss, D., Wright, J. F. and Furse, M. T. (1983). The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running-water sites. *Water Research* 17(3): 333-347.

Azrina, M. Z., Yap, C. K., Rahim Ismail, A., Ismail, A. and Tan, S. G. (2006). Anthropogenic impacts on the distribution and biodiversity of benthic macroinvertebrates and water quality of the langat river, peninsular malaysia. *Ecotoxicology and Environmental Safety* 64(3): 337-347.

Bakker, V. J. and Kelt, D. A. (2000). Scale-depandent patterns in body size distributions of neotropical mammals. *Ecology* 81(12): 3530-3547.

Barbour, M. T. and Yoder, C. O. (2000). The multimetric approach to bioassessment, as used in the

united states of america. In assessing the biological quality of fresh waters: Rivpacs

and other techniques (wright, j. F., sutcliffe, d. W. & furse, m. T., eds), . *Freshwater Biological Association, Ambleside, Cumbria, UK.*

: 281–292.

Bass, J. (1998). *Last instar larvae and pupae of the simulidae of britain and ireland: A key with brief ecological notes.* Institute of freshwater ecology.

Blackburn, T. M. and Gaston, K. J. (1994). Animal body size distributions: Patterns, mechanisms and implications. *Trends in Ecology & Evolution* 9(12): 471-474.

Blackburn, T. M. and Gaston, K. J. (1999).The relationship between animal abundance and body size: A review of the mechanisms. In *Advances in ecological research*, Vol. Volume 28, 181-210 (Eds A. H. Fitter and D. Raffaelli). Academic Press.

Bogatov, V. and Bogatova, L. (2009). Heavy metal accumulation by freshwater hydrobionts in a mining area in the south of the russian far east. *Russian Journal of Ecology* 40(3): 187-193.

Boisclair, D. and Leggett, W. C. (1985). Rates of food exploitation by littoral fishes in a mesotrophic north-temperate lake. *Canadian Journal of Fisheries and Aquatic Sciences* 42: 556-566.

Brown, J. H., Gillooly, J. F., Allen, A. P., Savage, V. M. and West, G. B. (2004). Towards a metabolic theory of ecology. *Ecology* 85: 1771-1789.

Brown, J. H., Marquet, P. A. and Taper, M. L. (1993). Evolution of body size: Consequences of an energetic definition of fitness. *The American Naturalist* 142(4): 573.

Burkholder, J. M., Noga, E. J., Hobbs, C. H. and Glasgow, H. B. (1992). New "Plantom" Dinoflagellate is the causative agent of major estuarine fish kills. *Nature* 358: 407-410.

Buss, D. and Salles, F. (2007). Using baetidae species as biological indicators of environmental degradation in a brazilian river basin. *Environmental Monitoring and Assessment* 130(1): 365-372.

Cain, J. R., Lawrence Klotz, R., Trainor, F. R. and Costello, R. (1979). Algal assay and chemical analysis: A comparative study of water quality assessment techniques in a polluted river. *Environmental Pollution (1970)* 19(3): 215-224.

Cassey, P. and Blackburn, T. M. (2004). Body size trends in a holocene island bird assemblage. *Ecography* 27(1): 59-67.

Chatelliers, M. C. D. and Reygrobellet, J. L. (1990). Interactions between geomorphological processes, benthic and hyporheic communities: First results on a by-passed canal of the french upper rhône river. *Regulated Rivers: Research & Management* 5(2): 139-158.

Cooper, A. H. and Waltham, A. C. (1999). Subsidence caused by gypsum dissolution at ripon, north yorkshire. *Quarterly Journal of Engineering Geology and Hydrogeology* 32(4): 305-310.

Cosser, P. R. (1989). Water quality, sediments and the macroinvertebrate community of residential canal estates in south-east queensland, australia: A multivariate analysis. *Water Research* 23(9): 1087-1097.

Croft, P. S. (1986). *A key to the maor groups of british freshwater invertebrates.*

Cyr, H., Peters, R. H. and Downing, J. A. (1997). Population density and community size structure: Comparison of aquatic and terrestrial systems. *Oikos* 80(1): 139-149.

Damuth, J. D. (1992). Taxon- free characterization of animal communities. . *In:Terrestrial Ecosystems Through Time Evolutionary Paleoecology of Terrestial plants and Animals(eds Behrensmeyer, A.K., Damuth,J.D., DiMichele,W.A., Potts,R.,Sues,H&Wing,S.L.).University of Chicago Press,Chicago,IL,USA.* : 183 -203.

Das, R., Samal, N. R., Roy, P. K. and Mitra, D. (2006). Role of electrical conductivity as an indicator of pollution in shallow lakes. *Asian Journal of Water, Environment and Pollution* 3(1): 143-146.

De'ath, G. and Fabricius, K. (2010). Water quality as a regional driver of coral biodiversity and macroalgae on the great barrier reef. *Ecological Applications* 20(3): 840-850.

Defra, U. (2010). Sensitive areas (eutrophic): Http://webcache.Googleusercontent.Com/search?Q=cache:Lo8hy-uhl1ej:Www.Defra.Gov.Uk/environment/quality/water/waterquality/sewage/sensarea/eutrophic-nitrates.Htm+river+foss+is+eutrophic+sensitive&cd=1&hl=en&ct=clnk&gl=uk.

Denby, L. and Mallows, C. (2009). Variations on the histogram. *Journal of Computational and Graphical Statistics* 18(1): 21-31.

Duplisea, D. E. and Drgas, A. (1999). Sensitivity of a benthic, metazoan, biomass size spectrum to differences in sediment granulometry. *Marine Ecology Progress Series* 177: 73-81.

Efron, B. and Tibshirani, R. J. (1993). *An introduction to the bootstrap.* Chapman and Hall, London, UK.

Egan, H. (1976). Chemical analysis and water quality. *Fresenius' Journal of Analytical Chemistry* 282(4): 323-329.

Elliott, J. M. H., U. H. and Macan, T. T (1988). *Larvae of the british ephemeroptera: A key with ecological notes.*: *Scientific Publications of the Freshwater Biological Association*.

Epler, J. H. (2006). *Identification manual for the aquatic and semi-aquatic heteroptera of florida.*: Division of WaterResources Management, Department of EnvironmentalProtection, State of Florida, Tallahassee, FL.

Feio, M. J., Almeida, S. F. P., Craveiro, S. C. and Calado, A. J. (2009). A comparison between biotic indices and predictive models in stream water quality assessment based on benthic diatom communities. *Ecological Indicators* 9(3): 497-507.

Feller, R. J. and Warwick, R. M. (1988). *Energeticsenergetics. In: Introduction to the study of meiofauna. Smithsonian institution press, washington,.* Smithsonian Institution, Washington DC.

Fu, C., Wu, J., Wang, X., Lei, G. and Chen, J. (2004). Patterns of diversity, altitudinal range and body size among freshwater fishes in the yangtze river basin, china. *Global Ecology & Biogeography* 13(6): 543-552.

Gaedke, U. (1992). The size distribution of plankton biomass in a large lake and its seasonal variability. *Limnology and Oceanography* 37(6): 1202-1220.

Gardezi, T. and Silva, J. d. (1999). Diversity in relation to body size in mammals: A comparative study. *The American Naturalist* 153(1): 110-123.

Garland, J. A. (1981). Observations on survival of eggs of hemerobius stigma (neuroptera: Hemerobiidae) following exposure to frost. . *Manitoba Ent* 12: 61-62.

Gaston, K. J. and Blackburn, T. M. (1996). Range size-body size relationships: Evidence of scale dependence. *Oikos* 75(3): 479-485.

Gaston, K. J., Chown, S. L. and Mercer, R. D. (2001). The animal species-body size distribution of marion island. *Proceedings of the National Academy of Sciences of the United States of America* 98(25): 14493-14496.

Gaston, K. J. and McArdle, B. H. (1994). The temporal variability of animal abundances: Measures, methods and patterns. *Philosophical Transactions: Biological Sciences* 345(1314): 335-358.

Giere, O. (1993). *Meiobenthology: The microscopic fauna in aquatic sediments* Springer-Verlag, Berlin.

Gooderham, J. and Tsyrlin, E. (2002). *The waterbug book: A guide to the freshwater macroinvertebrates of temperate australia.* CSIRO Publishing.

Grantham, B. A. and Hann, B. J. (1994). Leeches (annelida: Hirudinea)in the experimental lakes area, northwestern ontario, canada:Patterns of species composition in relation to environment. Canadian. *Journal of Fisheries and Aquatic Sciences* 5: 1600-1607.

Gray, N. F. (1999). *Water quality assessment in: Water technology: An introduction for environmental scientists and engineers* London: Elaine Wigzell.

Gutiérrez, J. and Iribarne, O. (2004). Conditional responses of organisms to habitat structure: An example from intertidal mudflats. *Oecologia* 139(4): 572-582.

Haines, T. A. (1981). Acidic precipitation and its consequences for aquatic ecosystems: A review. *Transactions of the American Fisheries Society* 110(6): 669-707.

Hall, D. J., Cooper, W. E. and Werner, E. E. (1970). An experimental approach to the production dynamics and structure of freshwater animal communities. *Limnology and Oceanography* 15(6): 839-928.

Hall, P. and York, M. (2001). On the calibraion of silvermans's test for multimodality. *Statistica Sinica* 11: 515-536.

Hammerton, D. (Ed) (1995). *Water pollution. In ralph c. Kirkwood, anita j.Longley. Clean technology and environment.*

Hanson, J. M. (1990). Macroinvertebrate size-distributions of two contrasting freshwater macrophyte communities. *Freshwater Biology* 24(3): 481-491.

Hanson, J. M., E.E.Prepas and W.C.Mackay (1989a). Size distribution of macro-invertebrates communities in fresh water lake. *Canaedian Journal of fishries and aquatic science* 46: 1510-1519.

Hanson, J. M., Prepas, E. E. and Mackay, W. C. (1989b). Size distribution of macro-invertebrates communities in fresh water lake

Canadian Journal of fishries and aquatic science 46: 1510-1519.

Havlicek, T. D. and Carpenter, S. R. (2001). Pelagic species size distributions in lakes: Are they discontinuous? *Limnology and Oceanography* 46(5): 1021-1033.

Hawkes, H. A. (1998). Origin and development of the biological monitoring working party score system. *Water Research* 32(3): 964-968.

Hawkins, C. P. (2006). Maintaining and restoring the ecological integrity of freshwater ecosystems: Refining biological assessments. *Ecol.Appl* 16: 1249.

Hem, J. D. (Ed) (1985). *Study and interpretation of the chemical characteristics of natural water.*: US Govt Printing Office, Washington.

Holling, C. S. (1992). Cross-scale morphology, geometry, and dynamics of ecosystems. *Ecological Monographs* 62(4): 447-502.

Holling, C. S. and Allen, C. R. (2002). Adaptive inference for distinguishing credible from incredible patterns in nature. *Ecosystems* 5(4): 319-328.

Howard, S. (2002).A guide to monitoring the ecological quality of ponds and canals using psym. Pond Action (Pond Conservation Trust: Policy and Research) Environment Agency Midland Region.

Hubbell, S. P. (1997). A unified theory of biogeography and relative species abundance and its application to tropical rain forests and coral reefs. *Coral Reefs* 16(5): S9-S21.

Hughes, B. D. (1978). The influence of factors other than pollution on the value of shannon's diversity index for benthic macro-invertebrates in streams. *Water Research* 12(5): 359-364.

Hünken, A. and Mutz, M. (2007). On the ecology of the filter-feeding neureclipsis bimaculata (trichoptera, polycentropodidae) in an acid and iron rich post-mining stream. *Hydrobiologia* 592(1): 135-150.

Hutchinson, G. E. (1959). Homage to santa rosalia, or why are there so many kinds of animals? *Am.Nat.,* 93: 245-249.

Hutchinson, G. E. and MacArthur, R. H. (1959). A theoretical ecological model of size distributions among species of animals. *The American Naturalist* 93(869): 117-125.

Ibanez, F. and Dauvin, J. C. (1988). Long-term changes (1977 to 1987) in a muddy fine sand abra alba - melinna palmata community from the western english channel: Multivariate time series analysis. *Marine Ecology Progress Series* 49: 65-81.

ISO (1979). Assessment of the biological quality of rivers by a macroinvertebrates 'score'. *ISO/TC 147/SC 5/WG 6 N 5*: 18.

James, K. L. and Joseph, V. M. (2003).In *Encyclopedia of Insecrs* (Eds V. H. Resh and R.T.Crade). Academic press.

James, M. R., Weatherhead, M., Stanger, C. and Graynoth, E. (1998). Macroinvertebrate distribution in the littoral zone of lake coleridge, south island, new zealand-effects of habitat stability, wind exposure, and macrophytes. *New Zealand Journal of Marine and Freshwater Research* 32(2): 287 - 305.

Jason, H. K. and Page, L. M. (2003). The evolution of body size in extant groups of north american freshwater fishes: Speciation, size distributions, and cope's rule. *The American Naturalist* 161(3): 413-421.

Kamenir, Y., Dubinsky, Z. and Zohary, T. (2004). Phytoplankton size structure stability in a meso-eutrophic subtropical lake. *Hydrobiologia* 520(1): 89-104.

Karr, J. R. (1996). *Rivers as sentinels: Using the biology of rivers to guide landscape managment. Pp in naiman and bilby, eds. The ecology and managment of streams and rivers in the pacific northwest coastal ecoregion. .* Springer -Verlag, New York.

Kerans, B. L., Karr, J. R. and Ahlstedt, S. A. (1992). Aquatic invertebrate assemblages: Spatial and temporal differences among sampling protocols. *Journal of the North American Benthological Society* 11(4): 377-390.

Kidwell, S. M. (2002). Mesh-size effects on the ecological fidelity of death assemblages: A meta-analysis of molluscan live-dead studies. *Geobios* 35(Supplement 1): 107-119.

Klemm, D. J. (1972). The leeches (annelida: Hirudinea) of michigan. *Michigan Academician* 4: 405-444.

Klimaszewski, J. and Kevan, D. K. (1985). The brown lacewing flies of canada and alaska (neuroptera:Hemerobiidae). Part 1. The genus hemerobius linnaeus: Systematics, bionomics and distribution. Macdonald college, mcgill university, lyman entomological museum and research laboratory. *Ste Anne de Bellevue, Quebec.*

Kolkwitz, R. and Marsson, M. (1902). Grundsätze für die biologische beurtheilung des wassers nach seiner flora und fauna, mittheilungen aus der königlichen prüfungsanstalt für wasserversorgung und abwässerbeseitung zu berlin. (1): 33-72.

Kosek, M. B., C. Guerrant, R.L (2003). The global burden of diarrhoeal disease, as estimated from studies published between 1992 and 2000. *Bull World Health Organ* 81: 197-204.

Kozlowski, J. (1996). Optimal initial size and adult size of animals: Consequences for macroevolution and community structure. *The American Naturalist* 147(1): 101-114.

Krawchuk, M. A. and Taylor, P. D. (2003). Changing importance of habitat structure across multiple spatial scales for three species of insects. *Oikos* (103): 153-161.

Lawrence, J. F., Hastings, A. M., Dallwitz, M. J., Paine, T. A. and Zurcher, E. J. (2000). Elateriformia(coleoptera): Descriptions, illustrations, identification, and information retrieve for families and subfamilies.

Leaper, R., Raffaelli, D., Emes, C. and Manly, B. (2001). Constraints on body-size distributions: An experimental test of the habitat architecture hypothesis. *Journal of Animal Ecology* 70(2): 248-259.

Lenat, D. R. (1988). Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. *Journal of the North American Benthological Society* 7(3): 222-233.

Lewis, D. R., Williams, R. J. and Whitehead, P. G. (1997). Quality simulation along rivers (quasar): An application to the yorkshire ouse. *Science of The Total Environment* 194-195: 399-418.

Littlewood, I. G., Watts, C. D. and Custance, J. M. (1998). Systematic application of united kingdom river flow and quality databases for estimating annual river mass loads (1975-1994). *Science of The Total Environment* 210-211: 21-40.

Logan, P. (2001). Ecological quality assessment of rivers and integrated catchment management in england and wales. *J. Limnol.* 60: 25-32.

Lovegrove, B. G. and Haines, L. (2004). The evolution of placental mammal body sizes: Evolutionary history, form, and function. *Oecologia* 138(1): 13-27.

MacArthur, R. H. and MacArthur, J. W. (1961). On bird species diversity. *Ecology* 42(3): 594-598.

Mackie, G. L. (2001). *Applied aquatic ecosystem concepts.* Kendall/Hunt Publishing Company.

Manly, B. F. J. (1996). Are there clumps in body-size distributions? *Ecology* 77(1): 81-86.

Marquet, P. A. and Cofre, H. (1999). Large temporal and spatial scales in the structure of mammalian assemblages in south america: A macroecological approach. *Oikos* 85: 299–309. .

Marquet, P. A., Navarrete, S. A. and Castilla, J. C. (1995). Body size, population density, and the energetic equivalence rule. *Journal of Animal Ecology* 64(3): 325-332.

Marquet, P. A., Quinones, R. A., Abades, S., Labra, F., Tognelli, M., Arim, M. and Rivadeneira, M. (2005). Scaling and power-laws in ecological systems. *J Exp Biol* 208(9): 1749-1769.

Mason, C. F. (Ed) (2002a). *Biology of fresh water pollution.*: Pearson Educated Limited.

Mason, C. F. (2002b). *Biology of fresh water pollution.*: Pearson Educated Limited.

Matyziak, K. (1979). Structure of leech groups (hirudineai)n polluted parts of the catchment area of the rivers bzura and ner.Part 1. Field investigations. *Acta Hydrobiologica (Krakow)* 18: 259-276.

Mau, H. E. L., Foottit, R. G., Hamilton, K. G. A. and Scudder, G. G. E. (2000). Checklist of the hemiptera of canada and alaska.

Maurer, B. A. and Brown, J. H. (1988). Distribution of energy use and biomass among species of north american terrestrial birds. *Ecology* 69(6): 1923.

McAbendroth, L., Ramsay, P. M., Foggo, A., Rundle, S. D. and Bilton, D. T. (2005). Does macrophyte fractal complexity drive invertebrate diversity, biomass and body size distributions? *Oikos* 111(2): 279-290.

McDonald, B., W. Borden and Lathrop, J. (1990).Citizen stream monitoring: A manual for illinois. Illinois Department of Energy and Natural Resources, ILENR/RE-WR-90/18. Springfield, Illinois

McKinney, M. L. (1997). Extinction vulnerability and selectivity: Combining ecological and paleontological views. *Annual Review of Ecology and Systematics* 28(ArticleType: primary_article / Full publication date: 1997 / Copyright © 1997 Annual Reviews): 495-516.

McNamee, P. J. (1979). A process model for eastern black-headed budworm. *Can. Ent.* 111: 55-66.

Metcalfe-Smith, J. L. (Ed) (1996). *Biological water quality assessment of running water based on macro-invertebrates communities.In petts, g and calow,p.(eds) river restoration,* . Blackwell, Oxford.

Metcalfe, J. L. (1989). Biological water quality assessment of running waters based on macroinvertebrate communities: History and present status in europe. *Environmental Pollution* 60(1-2): 101-139.

Michael, D. R. and Leland, J. J. (2005). The influence of habitat complexity on littoral invertebrate distributions: Patterns differ in shallow prairie lakes with and without fish. *Canadian Journal of Fisheries and Aquatic Sciences* 62(9).

Morand, S. and Poulin, R. (2002). Body size-density relationships and species diversity in parasitic nematodes: Patterns and likely processes. *Evol. Ecol. Res.* 4: 951–961.

Morin, A. and Nadon, D. (1991). Size distribution of epilithic lotic invertebrates and implications for community metabolism. *Journal of the North American Benthological Society* 10(3): 300-308.

Morse, D. R., Stork, N. E. and Lawton, J. H. (1988). Species number, species abundance and body length relationships of arboreal beetles in bornean lowland rain forest trees. *Ecological Entomology* 13(1): 25-37.

Munn, M. and King, R. (1987). Ecology of potamanthus myops (walsh) (ephemeroptera: Potamanthidae) in a michigan stream (USA)

journal hydrobiologia *Hydrobiologia* 146: 71-75.

Nummi, P., Sjoberg, K., Poysa, H. and Elmberg, J. (2000). Individual foraging behaviour indicates resource limitation: An experiment with mallard duckling. *Can.J.Zool.* 78: 1891-1895.

Oksanen, L., Fretwell, S. D. and Jarvinen, O. (1979). Interspecific aggression and the limiting similarity of close competitors: The problem of size gaps in some community arrays. *The American Naturalist* 114(1): 117-129.

Paine, R. T. (1966). Food web complexity and species diversity. *The American Naturalist* 100(910): 65-75.

Pantle, R. and Buck, H. (1955). Die biologische uberwachung der gawasser und die darstellung der ergebnisse. Gas-u- wasserfach. 96.

Parry, D. M., Kendall, M. A., Rowden, A. A. and Widdicombe, S. (1999). Species body size distribution patterns of marine benthic macrofauna assemblages from contrasting sediment types. *Journal of the Marine Biological Association of the UK* 79(05): 793-801.

Pechman, J. H. K., ScottT, D. E., Semlitsch, R. D., Calwell, J. P., VITT, L. J. and Gibbons, J. W. (1991). Declining amphibian populations: The problem of separating human impacts from natural fluctuations. *Science* 253(5022): 892-895.

Perrin, N. (1998). On body size, and energy and fitness. *Functional Ecology* 12(3): 500-502.

Peter, H. and Matthew, Y. (2001). On the calibration of silverman's test for multimodality. *Statistica Sinica* 11: 515-536.

Peters, R. (1983a). Size stucture of the plankton community along the tropjic gradient of lake memphremagog. *Can J Fish Aquat Sci* 40: 1770-1778

Peters, R. H. (1983b). The ecological implication of body size. *Cambridge University Press,Cambridge*.

Pinder, L. C. V. and Farr, I. S. (1987). Biological surveillance of water quality. Temporal and spatial variation in the macro-invertebrates fauna of the river frome, a dorset chalk stream. *Arch. Hydrobiology.,* 109: 321-331.

Pinder, L. C. V., Ladle, M., Gledhill, T., Bass, J. A. B. and Matthews, A. M. (1987). Biological surveillance of water quality. A comparison of macroinvertebrates surveillance method in realtion to assessment of water quality, in a chalk streams. *Arch. Hydrobiologie* 109: 207-226.

Pittman, S. J., McAlpine, C. A. and Pittman, K. M. (2004).Linking fish and prawns to their environment: A hierarchical landscape approach. Inter-Research.

Platt, T. and Denman, K. (1978). The structure of pelagic marine ecosystem. *Rappor et proces-Verbaux Reunions.Conseil International pour , Explorarion de la Mer* 173: 60-65.

Poff, N. L., Palmer, M. A., Angermeier, P. L., Vadas, R. L., Jr., Hakenkamp, C. C., Bely, A., Arensburger, P. and Martin, A. P. (1993). Size structure of the metazoan community in a piedmont stream. *Oecologia* 95(2): 202-209.

Prazen, E. (1962). On the estimation of a probability density and mode. *Annals of Mathematics and Statistics* 33: 1065-1076.

Pyron, M. (1999). Relationships between geographical range size, body size, local abundance, and habitat breadth in north american suckers and sunfishes. *Journal of Biogeography* 26(3): 549-558.

R Core Development Team, R. (2004).A language and environment for statistical computing. In *R Foundation for Statistical Computing, Vienna, Austria* http://www.R-Project.org.

Raffaelli, D., Hall, S., Emes, C. and Manly, B. (2000). Constraints on body size distributions: An experimental approach using a small-scale system. *Oecologia* 122(3): 389-398.

Raffaelli, D. and Moller, H. (1999).Manipulative field experiments in animal ecology: Do they promise more than they can deliver? In *Advances in ecological research*, Vol. Volume 30, 299-338 (Eds A. H. Fitter and D. G. Raffaelli). Academic Press.

Raffaelli, D. G. and Frid, C. L. (Eds) (2010). *Ecosystem ecology: A new synthesis* Cambridge University Press

Rasmussen, J. (1993). Patterns in the size structure of littoral zone macroinvertebrates communites *Can J Fish Aquat Sci* 50: 2192-2207.

Rath, L. C. (1986).Le rapport entre heterogenite due benthos et les sediments lacustres dans les zones profondes et littoriprofodes. In *University de Montreal*, Vol. MSC thesis.

Reash, R. (1990). Physicochemical factors affecting the abundance and species richness of fishes in the cimarron river. *Proc. Okla. Acad. Sci* 70: 23-28.

Redfern, M. (1975). Revised field key to the invertebrate fauna of the stony hills streams 4: 105-115.

Restrepo, C., Renjifo, L. M. and Marples, P. (1997). Frugivorous birds in fragmented neotropical montane forests: Land scape pattern and body mass distribution. In: *Tropical forest remnants: Ecology managment and conservation of fragmented communities* (eds laurance, w.F, bierregaad, r. O. And moritz, c.). University of chicago press. *Chicago, IL, USA*: 171-189.

Richard, E., White, D. J. B. and Peterson, R. T. (1998). *A field guide to insects.* Houghton Mifflin Co

Robson, B. J., Barmuta, L. A. and Fairweather, P. G. (2005). Methodological and conceptual issues in the search for a relationship between animal body-size distributions and benthic habitat architecture. *Marine and Freshwater Research* 56(1): 1-11.

Rosenberg, R., Blomqvist, M., C Nilsson, H., Cederwall, H. and Dimming, A. (2004). Marine quality assessment by use of benthic species-abundance distributions: A proposed new protocol within the european union water framework directive. *Marine Pollution Bulletin* 49(9-10): 728-739.

Rosenblatt, M. (1956). Remarks on some nonparametric estimates of a density function. *The Annals of Mathematical Statistics* 27(3): 832-837.

Royama, T. (1984). Population dynamics of the spruce budworm choristoneura fumiferana. *Ecological Monographs* 54(4): 429-462.

Rutt, G. P., Pickering, T. D. and Reynolds, N. R. M. (1993). The impact of livestock farming on welsh streams: The development and testing of a rapid biological method for use in the assessment and control of oganic pollution from farms. *Environ.Pollution* 81: 217-228.

Sakamoto, M., Ogamino, Y. and Tanaka, Y. (2010). Leptodora kindtii: A cladoceran species highly sensitive to toxic chemicals. *Limnology* 11(2): 193-196.

Sawyer, R. T. (1986). *Leech biology and behavior.* Oxford University Press,

Oxford, U.K.

Sawyer, R. T. and Hammond, D. L. (1973). Observations on themarine leech calliobdella carolinensis (hirudinea: Piscicolidae), epizootic on the atlantic menhaden. *Biological Bulletin. Marine Biological Laboratory,Woods Hole* 145: 373-388.

Schmid, P. E., Tokeshi, M. and Schmid-Araya, J. M. (2002). Scaling in stream communities. *Proceedings. Biological sciences The Royal Society* 269(1509): 2587-2594.

Schwinghamer, P. (1981a). Characteristics size distribution of integrakl benthic communities. *Candian Journal of Fishries and Aquatic Science* 38: 1255-1263.

Schwinghamer, P. (1981b). Characteristics size distribution of integral benthic communities. *Candian Journal of Fishries and Aquatic Science* 38: 1255-1263.

Schwinghamer, P. (1983). Generating ecological hypotheses from biomass spectra using causal analysis: Abenthic example. *Marine Ecology Progress Series* 13: 151-166.

Schwinghamer, P. (1985). *Observations on size sructure and pelagic coupling of some shelf and abyssal benthic communities. In proceeding of nineteenth european marine biology symposium (ed.P.E.Gibbs), pp. 371-380.* Cambridge University Press, Cambridge.

Schwinghamer, P. (1988). Influence of pollution along a natural gradient and in mesocosm experiment on bomass- size spectra of benthic communities. *Mar. Ecol- Prog Ser.* 46:: 199-206.

Scmidt- Nielsen, K. (1984). *Scaling- why is animal size so important?* . Cambridge, Cambridge University press.

Seaman, D. E. and Powell, R. A. (1996). An evaluation of the accuracy of kernel density estimators for home range analysis. *Ecology* 77(7): 2075-2085.

Sendzimir, J., Allen, C. R., Gunderson, L. and Stow, C. A. (Eds) (2003). *Implications of body mass pattern: Linking ecological structure and processes to wildlife conservation and management. In: Landscape ecology and resource management: Linking theory with practice.*: Island press, Washington, DC, USA.

Sheldon, R. W., A.Prakash and W.H.Sutcliffe, J. (1972). The size distribution of particles in the oceaon. *Limnology and Oceanography* 17:: 327-340.

Sheldon, R. W. and Parsons, T. R. (1967a). A continuous size spectrum for particulate matter in the sea. *J. Fish. Res. Board Canada.* 24: 909-915.

Sheldon, R. W. and Parsons, T. R. (1967b). A continuous size spectrum for particulate matter in the sea. *J. Fish. Res. Board Canada.* 24: 909–915.

Siemann, E. and Brown, J. H. (1999). Gaps in mammalian body size distributions reexamined. *Ecology* 80(8): 2788-2792.

Silva, M., Brimacombe, M. and Downing, J. A. (2001). Effects of body mass, climate, geography, and census area on population density of terrestrial mammals. *Global Ecology and Biogeography* 10(5): 469-485.

Silverman, B. W. (1981). Using kernel density estimates to investigate multimodality. *Journal of the Royal Statistical Society. Series B (Methodological)* 43(1): 97-99.

Silvert, W. (1996). Size-aggregation in models of aquatic ecosystems. *Science of The Total Environment* 183(1-2): 107-114.

Singer, R. (1982). *Effects of acidic precipitation on benthos. Pages 329-363 in f.M. D'ltri(editor). Acidic precipitation: Effects on ecological system.* . Ann Arbor Scientific Publisher, Ann Arbor, Michigan.

Singh, A. and Gaur, J. (1989). Algal epilithon and water quality of a stream receiving oil refinery effluent. *Hydrobiologia* 184(3): 193-199-199.

Smith, D. G. (2001). *Annelida. Pennak's freshwater invertebrates of the united states: Porifera to crustacea.* John Wiley & Sons.

Smith, F. A., Brown, J. H., Haskell, J. P., Lyons, S. K., Alroy, J., Charnov, E. L., Dayan, T., Enquist, B. J., Ernest, S. K. M., Hadly, E. A., Jones, K. E., Kaufman, D. M., Marquet, P. A., Maurer, B. A., Niklas, K. J., Porter, W. P., Tiffney, B. and Willig, M. R. (2004). Similarity of mammalian body size across the taxonomic hierarchy and across space and time. *The American Naturalist* 163(5): 672-691.

Smock, L. A. (1980). Relationships between body size and biomass of aquatic insects. *Freshwater Biology* 10(4): 375-383.

Solimini, A. G., Benvenuti, A., D'Olimpio, R., Cicco, M. D. and Carchini, G. (2001). Size structure of benthic invertebrate assemblages in a mediterranean river. *Journal of the North American Benthological Society* 20(3): 421-431.

Soos, A. (1968). Identification key to the species of the genuserpobdellad e blainville1, 818 (hirudinoidea:Erpobdellidae). *Annales Historico-Naturales Musei Nationalis Hungarici* 60: 141-145.

Sorrell, S. (2001). The meaning of batneec: Interpreting excessive costs in uk industrial pollution regulation.

Sriyaraj, K. and Shutes, R. B. E. (2001). An assessment of the impact of motorway runoff on a pond, wetland and stream. *Environment International* 26(5-6): 433-439.

Stanley, S. M. (1986). Population size, extinction, and speciation; the fission effect in neogene bivalvia. *Paleobiology* 12(1): 89-110.

Stark, J. D. (1993). Performance of the macroinvertebrate community index: Effects of sampling method, sample replication, water depth, current velocity, and substratum on index values. *New Zealand Journal of Marine and Freshwater Research* 27(4).

Stead, T. K., Schmid-Araya, J. M. and Hildrew, A. G. (2003). All creatures great and small: Patterns in the stream benthos across a wide range of metazoan body size. *Freshwater Biology* 48(3): 532-547.

Strayer, D. (1986). The size structure of a lacustrine zoobenthic community. *Oecologia* 69(4): 513-516.

Strayer, D. L. (1991). Perspectives on the size structure of lacustrine zoobenthos, its causes, and its consequences. *Journal of the North American Benthological Society* 10(2): 210-221.

Stubblefield, J. W., Seger, J., Wenzel, J. W. and Heisler, M. M. (1993). Temporal, spatial, sex-ratio and body-size heterogeneity of prey species taken by the beewolf philanthus sanbornii (hymenoptera: Sphecidae). *Philosophical Transactions: Biological Sciences* 339(1290): 397-423.

Symonds, M. R. E. (1999). Insectivore life histories: Further evidence against an optimum body size for mammals. *Functional Ecology* 13(4): 508-513.

Thiebaux, M. L. and Dickie, L. M. (1992). Models of aquatic biomass size spectra and the common structure of their solutions. *Journal of Theoretical Biology* 159(2): 147-161.

Towers, D. J., Henderson, I. M. and Veltman, C. J. (1994). Predicting dry weight of new zealand aquatic macroinvertebrates from linear dimensions. *New Zealand Journal of Marine and Freshwater Research* 28(2): 159 - 166.

Vanaverbeke, J., Steyaert, M., Vanreusel, A. and Vincx, M. (2003). Nematode biomass spectra as descriptors of functional changes due to human and natural impact. *Marine Ecology Progress Series* 249: 157-170.

Vermaat, J. E., Eppink, F., van den Bergh, J. C. J. M., Barendregt, A. and van Belle, J. (2005). Aggregation and the matching of scales in spatial economics and landscape ecology: Empirical evidence and prospects for integration. *Ecological Economics* 52(2): 229-237.

Vernberg, F. J. and Vernberg, W. B. (Eds) (1974). *Pollution and physiology of marine organisms* Academic Press, New York.

Vicente, J., Colmenarejo, M. F., Sánchez, E., Rubio, A., García, M. G., Borja, R. and Jimenez, A. M. (2009). Evaluation of the water quality in the guadarrama river at the section of las rozas-madrid, spain. *Water and Environment Journal*.

Vuori, K. M. P., M. (1996). Assessing pollution of the river kymijoki via hydropsychid caddis flies. *Archiv für Hydrobiologie* 136(2): 171-190.

Wade, S. C., T and Marie McDowell,L (2004). Critter catalogue. A guide to the aquatic invertebrates of south australian inland waters. Waterwatch south australia.

Walker, B. and Salt, D. (Eds) (2006). *Resilience thinking: Sustaining ecosystems and people in a changing world.*

Warwick, R. M. (1984). Species size distributions in marine benthic communities. *Oecologia* 61(1): 32-41.

Warwick, R. M. and Clarke, K. R. (1993). Comparing the severity of disturbance: A metaanalysis of marine macrobenthic community data. *Marine Ecology Progress Series* 92: 221-231.

Warwick, R. M. and Clarke, K. R. (1994). Relearning the abc: Taxonomic changes and abundance/biomass relationships in disturbed benthic communities. *Marine Biology* 118(4): 739-744.

Watson, L. and Dallwitz, M. J. (2003). British insects: The families of trichoptera. Version: 28th august 2009.

Insect groups treated separately in the ‘british insects’ package

Welker, C., De Negro, P. and Sarti, M. (2001). Green algal carotenoids and yellow pigmentation of rainbow trout fish. *Aquaculture International* 9(1): 87-93.

West, G. B., Brown, J. H. and Enquist, B. J. (1997). A general model for the origin of allometric scaling laws in biology. *Science* 276(5309): 122-126.

Whittle, P. (1958). On the smoothing of probability density functions. *Journal of the Royal Statistical Society. Series B (Methodological)* 20(2): 334-343.

Whitton, B. A. (1991). *Aims of monitoring. In: Whitton, b.A., rott, e., friedrich, g. (eds.), use of algae for monitoring rivers.*: Institut fur Botanik, Universitut Insbruck, Austria.

WHO (2004). Burden of disease and cost-effectiveness estimates.

Wiens, J. A. and Rotenberry, J. T. (1981). Habitat associations and community structure of birds in shrubsteppe environments. *Ecological Monographs* 51(1): 21-41.

William, H. R. (2005). *Handbook of urban insects and arachnids.* Cambridge university press.

Winner, R. W., Van Dyke, J. S., Caris, N. and Farrel, M. P. (1975). Response of the macroinvertebrate fauna to a copper gradient in an experimentally-polluted stream. . *Verh. Internat. Verein. Limnol.* 19: 2121–2127.

Wirwa. and Nicholas, L. (2008). Macroinvertebrates response to management strategies and habitat condition at kealia pond nwr, maui, hawaii.M.S. *South Dakota State University*.

Zamora-Muñoz, C., Sáinz-Cantero, C. E., Sánchez-Ortega, A. and Alba-Tercedor, J. (1995). Are biological indices bmpw' and aspt' and their significance regarding water quality seasonally dependent? Factors explaining their variations. *Water Research* 29(1): 285-290.