Research Article

The Variation of Size Distributions of Benthic Communities Across a Range of Irrigating Ponds and Canals of North Yorkshire, UK

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Abstract | The study on body size distribution of benthic creatures was carried out to assess the quality of irrigating water and to reveal the real environmental condition of ponds and canal system of North Yorkshire, UK. Samples of benthic communities from eight different stations were collected from the bed of ponds and canals with a pond net. The sites were varying in their qualities as determined in the laboratory by Average Score Per Texon (ASPT) as well as water conductivity. For most sites the body mass distributions were skewed towards smaller body sizes with another apparent mode for large size class. Contrary Kernel Density Estimates (KDE) for body mass spectra revealed marked differences in modalities among the sites. The distributions of body masses in cleanest and polluted sites were inversely related with water quality while intermediate quality sites were highly variable in modality and there is no consistency between the number of modes, ASPT and Biological Monitoring Working Party (BMWP) score. If anything, the negative relationship has been revealed between water quality (BMWP score and ASPT) and the number of modes and this negative correlation was significant R^2 = 0.5072, p <0.05.

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Introduction

Macro-invertebrates are bio-indicators whose presence or absence indicates environmental conditions of particular ecosystem. These living organisms interact with varying level of pollutants within their environment. Thus, in aquatic ecosystem these creatures can be used to assess the degree of quality on the basis of presence or absence of sensitive or tolerant species. The macro-invertebrate community in the littoral zone of static water 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. Biotic indices such as BMWPS and ASPT used for quality assessment of water, where indicator species are given a score on the basis of their sensitivity or tolerance to pollution. Sensitive species are given high score while tolerant species are given least score and the sum of the score provides an index of pollution for the sites (Mason, 2002). Tamiru (2019) found decling of sensitive species of macroinevrtebrates in impaired sites of river shinta, Gondar, Ethiopia. Body size distributions changed in benthic communities in response to nutrients in freshwater ponds (Hall *et al*., 1970) and in lakes ecosystem there are associations of macro-fauna with aquatic plants, which provide shelter for 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). The changing patterns in distributions of body mass of macro invertebrate with pollution levels were found in River Aire, UK (Bibi *et al*., 2019).

However, relatively little information is available on the size structure of freshwater benthic communities (Strayer, 1986; Hanson, 1990). Differences occur among lakes in macro-benthic 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 community in Mirror Lake, New Hampshire (Strayer, 1991), 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*., 1989; Morin and Nadon, 1991). The size and abundance of phytoplankton showed variability with nutrients concentration rather than with depth in the lake ecosystem (Baho *et al*., 2020). Thus, this study was carried out to find the body mass distributions of macro-invertebrates in a range of static and slow moving irrigating water bodies and to analyze the relationship of size based indicator for ecological resilience of these irrigating water courses using rigorous approach such as Kernel Density Estimates (KDE) (Tsuyuk *et al*., 2012).

Materials and Methods

The samples were collected from eight different static and slow moving habitats (ponds and canals) which includes Millington Pond, Coates Bridge (canal), Canal Head, Thornton Church Bridge (canal), Haggs Bridge (canal), Bielby Pond, Haggs Pond and Barmby Pond (Figure 1). The macro-invertebrates were collected using a pond net fitted with a 200 µm pore size with a frame height of 200 mm and width of 300 mm, attached with 1.5m handle. In this method the net was pushed for 5 meters through the top of few cm of substrate (mud) and the large area covered. The 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 benthic creatures were separated from debris using 250µm mesh sieve. Individuals were identified to the lowest taxonomic level with the help of a range of identification keys (Croft, 1986; Bass, 1998; Elliott, 1988) at least to family level, and counted for each sample.

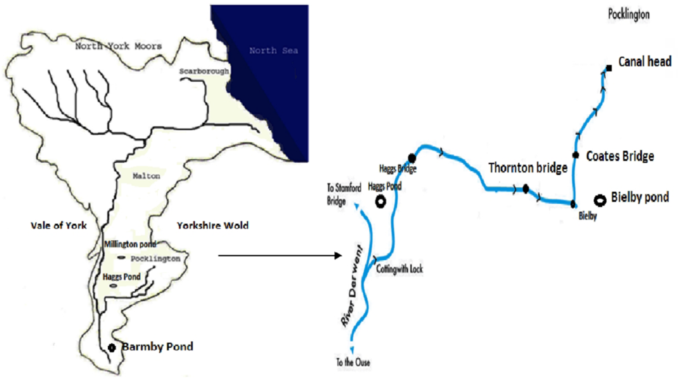

Figure 1: *The study sites of ponds and canals.*

Biological indices, such as Community Abundance, BMWPs assigned to the taxa (Table 1) and 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. 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. 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 approximate (Table 2) 2-D or 3-D geometric shapes and water displacement technique were applied as appropriate to derive the dry body mass of taxonomic groups (Stead *et al*., 2003; Feller and Warwick, 1988;

Table 1: *Biological Monitoring Working Party (BMWP) scores assigned for fresh water benthic communities by Hawkes (1998).*

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

Leaper *et al.*, 2001; Smock, 1980; Towers *et al.*, 1994). Size distributions were analyzed by frequency histogram and Kernel Density Estimation (KDE) (Kirsty *et al*, 2014) which determines the clumps for similar body mass across different taxa. A 5 % significance level was applied to the estimate of the density function underlying a body mass distribution.

Results and Discussion

Taxon richness

A total of 31 families of invertebrates were identified at these water courses (Table 3). 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, the number 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 taxa of stream (flowing) environments. Millington Pond is dominated by Chironomidae, Gammaridae, Asellidae Hydrobiidae and Beetle Larvae (Haliplidae), which are considered comparatively sensitive taxa to pollution. 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 while

Table 2: *Allometric 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	References
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) lnDM(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)$^{0.8205}$	
	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)	
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	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 In-L(HW-DW) converted BL to HW by using HW:BL	(Tower *et al*., 1994)

Corophidae, Hydrobiidae, Asellidae and Sphaeriidae, made a large proportion of the fauna at Haggs Bridge. Although the numbers of individuals were very low at Bielby, the families of Asellidae and Polycentropidae were abundant at this site. At Haggs Pond, 78 percent of individuals were Physidae, while Chironomida made up a large proportion of individuals at Barmby Pond. These findings suggests that the gradient of water quality in Millington Pond is cleanest while Barmby Pond classified as poor quallity site due to species richness and diversity as these parameters have potential functional responses to increasing toxic pressure in stream (Archaimbault *et al*., 2010). The highest number of taxa was present at Millington Pond and Thronton Church Bridge, while the number of individuals highest at Canal head, dominated by Asellidae. Fewer taxa were recorded at Haggs Bridge and the lowest number of individuals 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 absent in all the 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 the absence of standarized assessment methodology, because these ecosystems are less monitored by regulatory agencies. 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 indices

The Biological Monitoring Working Party (BMWP) score is an approach used to determine the gradient of water quality and routinely practiced by Environmental Agency UK. The issue of habitat changes of macro-invertebrates in water while using biotic index can be solved by pooling of samples (Carter *et al*., 2017). The BMWP score (Figure 2) was highest for

Table 3: *The number of individuals recorded at diiferent sites of static water bodies.*

Taxa	Millington	Coates	Canal head	Church bridge	Haggs bridge	Bielby	Haggs Pond	Barmby
Ancylidae	1	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

Millington Pond (62) with lower values (25) and (26) at Haggs Bridge. The value decreases from 57 (Thornton Church Bridge) to 26 for Bielby Pond and none of the sites have a value greater than 100, normally associated with clean flowing water systems. 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 (Figure 3) and the value at Bielby are 3.72 while Haggs pond has almost same value as Barmby pond. There is, however, no consistency in BMWP and ASPT values for intermediate sites. Variable findings have reported by using BMWPs, ASPT and BMWP-CR in the quality assessment of water of River Aturukuku in eastern Uganda (Ochieng *et al.*, 2020). This is 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) but the findings of Roche *et al.* (2010) revealed that ASPT diminish the impact of season variability on organisms abundances on the indicted degree of pollution. The overall correlation seems good between the two techniques (Figure 4) $R^2 = 0.5757$, $p < 0.05$.

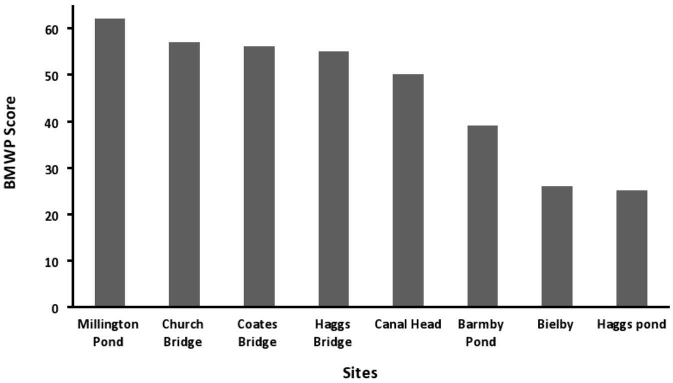

Figure 2: *Biological monitoring party (BMWP) scores.*

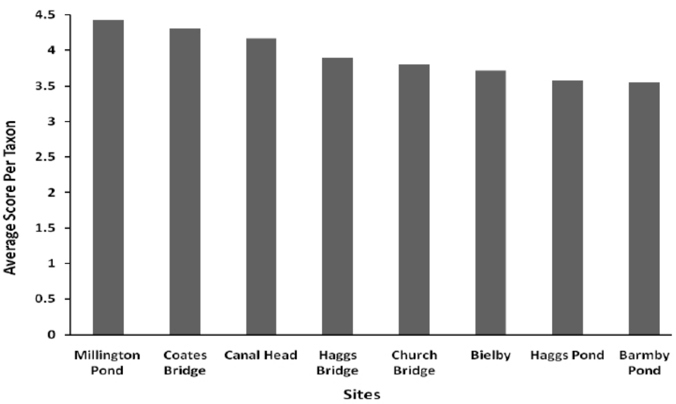

Figure 3: *Average score per Taxon (ASPT).*

Water conductivity

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 the value commonly found in natural water as a linear relationship has been found between electrical conductivity and level of pollution in lake water (Das *et al.*, 2006). 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 (Table 4). The Biological Indices 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, $R^2 = 0.5595$ $p<0.05$).

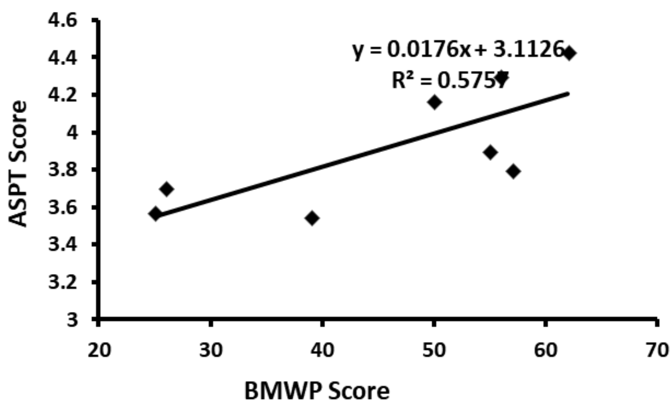

Figure 4: *The relationship between ASPT and BMWP at different sites of irrigating water bodies.*

Body size distributions in benthic communities

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 6). However visual inspections or frequency histogram of body size plots are unreliable (O'Sullivan and Rassel, 1995), due to possible bin size effects. Statistically rigorous estimates kernel density analysis of body mass spectra

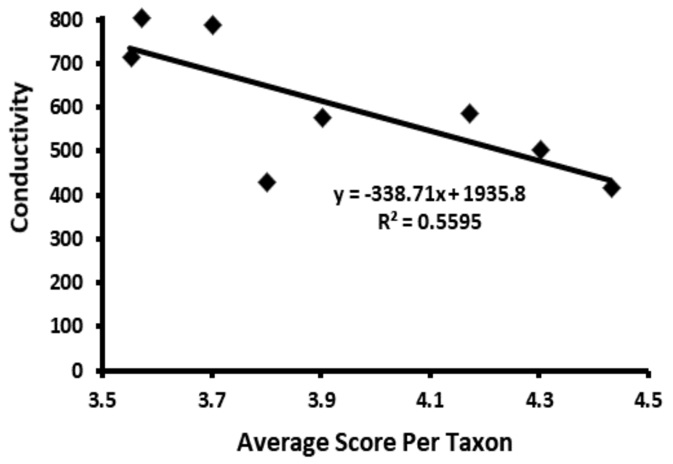

Figure 5: *The relationship between conductivity and ASPT for different irrigating water courses.*

Table 4: *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

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, Figure 7). The number of modes ranges from 1-4 for different static water bodies which 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 polluted and Barmby the cleaniest) because both biotic indices and the conductivity measurement do not support this inverse relationship. The size spectra for static water habitats have been reported as strikingly different from those in marine benthic communities (Strayer, 1986), but in this study it can be seen to be highly variable in the number of modes. Figure 7 shows that the number of modes increased with declining water quality. If anything, the negative relationship has been revealed between water quality (BMWP score and ASPT) and the number of modes (Figure 8) and this inverse correlation was significant R^2 =0.5072, $p < 0.05$).

Table 5: *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	

Conclusions and Recommendations

The evaluantion of water quality by chemical parameter represents regular monitorig approach which can be changed and inaccurate due to itermittant discharge of pollutants but the methods of biotic monitoring represents response based monitoring approach which provide long term information about the area therefore the study was carried out on size based indicators of freshwater benthic communities to find the real ecological condition of static and slow moving water of North Yorkshire, UK. The findings on static and slow moving water bodies showed variation in the number of modes in body size spectra of benthic communities, However size pattern does not appear to be related to water quality changes in these water courses In fact, the relationships seem to be the reverse. Therefore it is recommended that both chemical and biotic approches should be used in integrated way for monitoring purposes. It is also necessary to make adoptive changes in the indices and integrate them into multivariate approaches. It is further recommended that the similar experiment should be carriedout to arifically disturbed site with clean habiat.

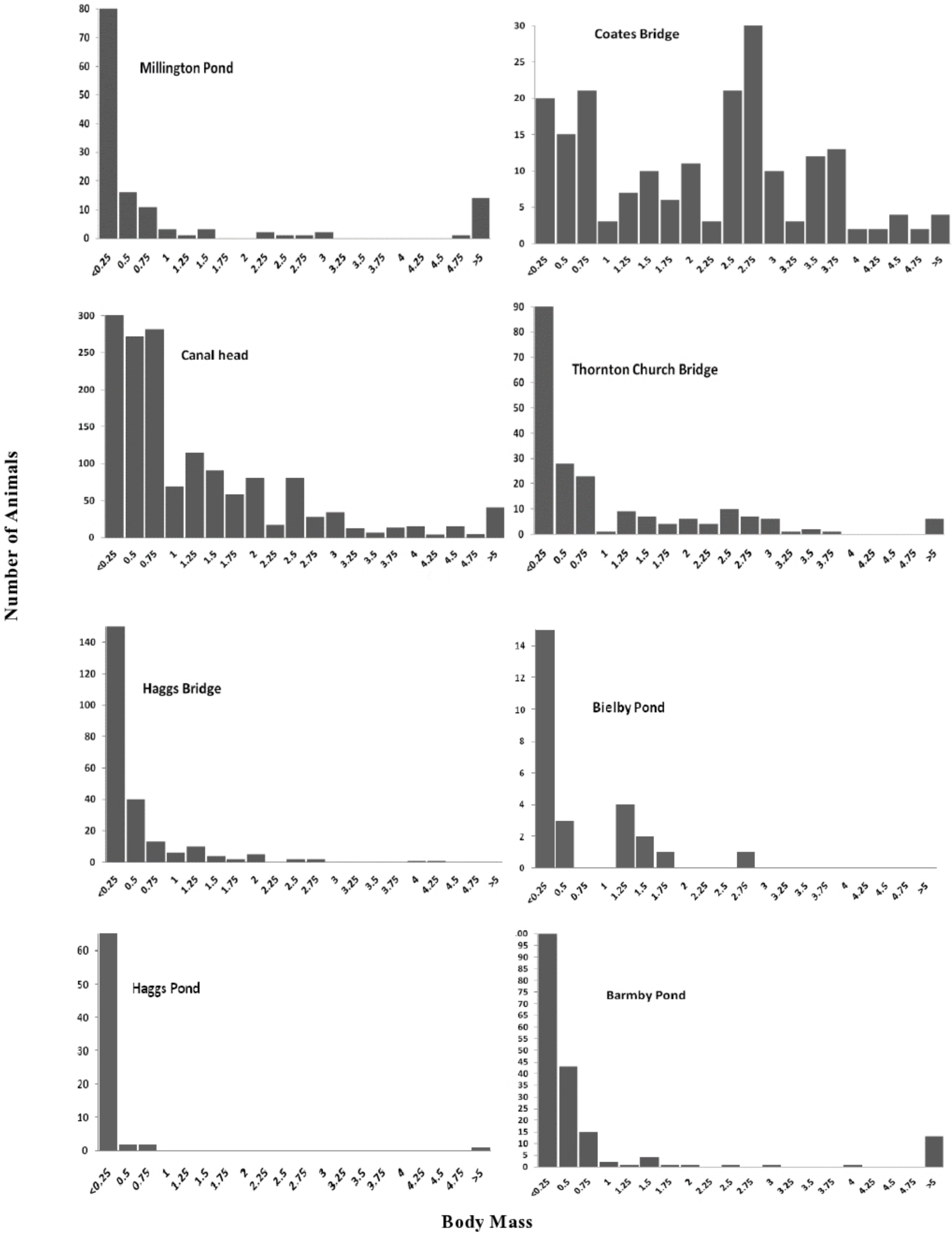

Figure 6: *Frequency histograms of body mass distributions.*

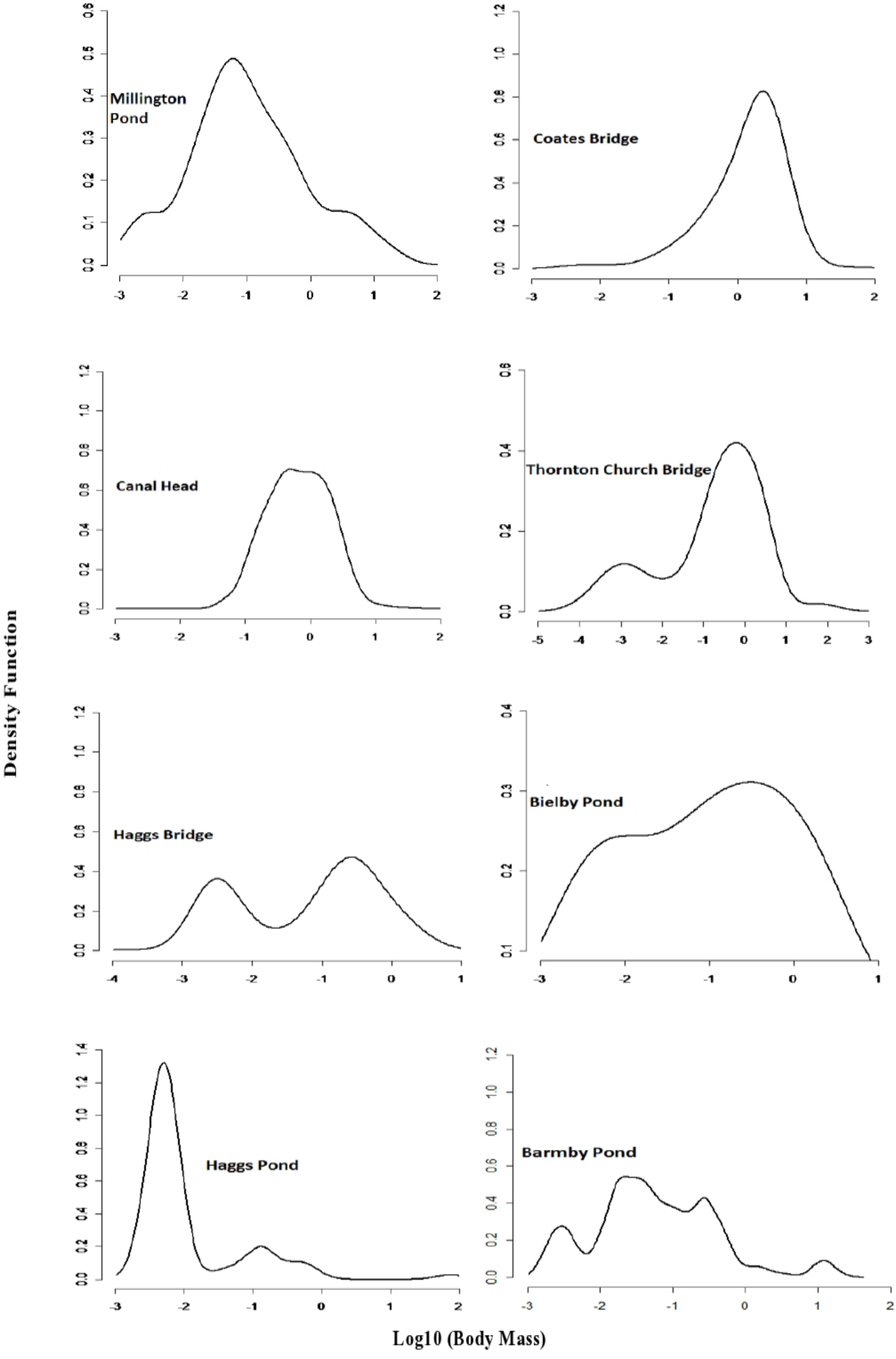

Figure 7: *The fitted density distributions for body sizes derived by kernel density estimation and bootstrapped re-sampling.*

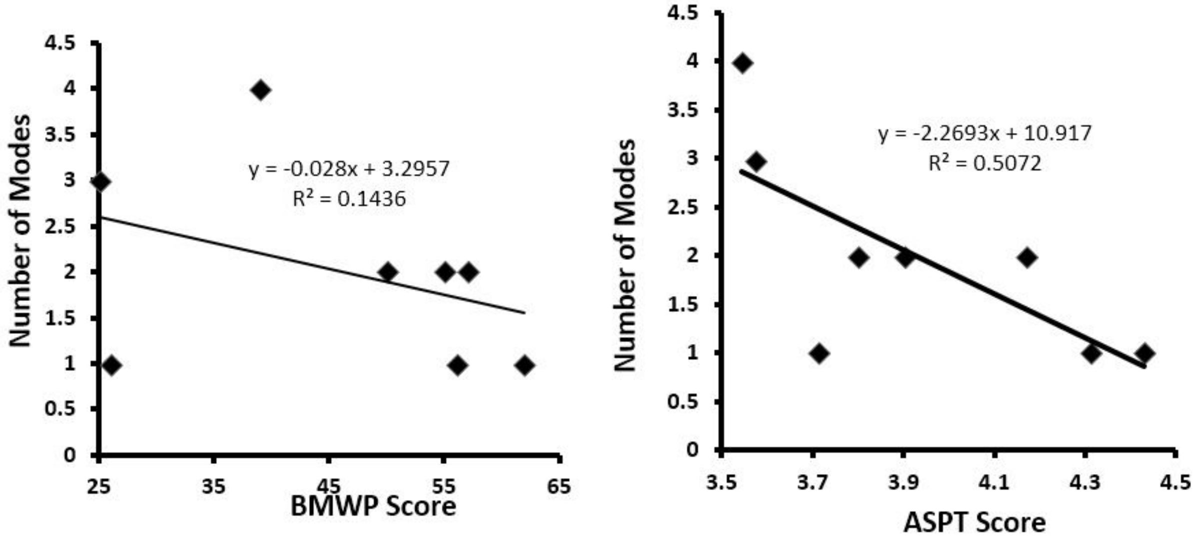

Figure 8: *The relationships between biotic indices (BMWP and ASPT) as water quality and number of modes (determined by Kernel Density Estimation (KDE).*

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Novelty Statement

The present research represents the environmental conditions and the resilience of ecological system of static and slow moving irrigating watercourses using biotic monitoring approaches and size based indicators which have been done previously.

Author's Contribution

Hamida Bibi: Did field and laboratory work, analyzed the results and wrote the manuscript.
Dave Rraffaelli: Superved the reaserch both in field and laboratry and proofread the manuscript.
Muhammad Sharif: Proofread and helped in revision of the manuscript.

Conflict of interest
The authors have declared no conflict of interest.

References

Archaimbault, V., P. Usseglio-Polatera, J. Garric, J. G.wasson and M. Babut. 2010. Assessing pollution of toxic sediment in streams using bio-ecological traits of benthic macroinvertebrates. Freshwat. Biol., 55(7): 1430-1446.

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.

Baho, D.L., D. Stina, K. Richard and D. Angeler. 2020. *Phytoplankton size- and abundance-based resilience assessments reveal nutrient rather than water level effects.* Sci. Total Environ., 746, 141110: 1-9. https://doi.org/10.1016/j.scitotenv.2020.141110

Boisclair, D. and W.C. Leggett. 1985. Rates of food exploitation by littoral fishes in a mesotrophic north-temperate lake. Can. J. Fish. Aquat. Sci., 42: 556-566. https://doi.org/10.1139/f85-073

Bibi, H., D. Raffaelli, M. Iqbal, M. Sharif and M.J.K. Khattak. 2019. Body Size Patterns in Stream Communities: A Test of Holling's Textural Discontinuity Hypothesis. J. Entomol. Res. Soc., 21(1): 17-35.

Cain, J.R., R. Lawrence Klotz, F.R. Trainor and R. Costello. 1979. Algal assay and chemical analysis: A comparative study of water quality assessment techniques in a polluted river. Environ. Pollut., (1970) 19(3): 215-224. https://doi.org/10.1016/0013-9327(79)90043-0

Chatelliers, M.C.D. and J.L. Reygrobellet. 1990. Interactions between geomorphological processes, benthic and hyporheic communities: First results on a by-passed canal of the french upper rhône river. Regulat. Rivers Res. Manage., 5(2): 139-158. https://doi.org/10.1002/rrr.3450050205

Carter, J.L., V.H. Resh and M.J. Hannaford. 2017. Macroinvertebrates as biotic indicator of environmental quality. In: Hauer, R and Lamberti, G., Eds., Methods in Stream Ecology, 3rd Edition, Academic press, UK, 293 - 318. https://doi.org/10.1016/B978-0-12-813047-6.00016-4

Cosser, P.R., 1989. Water quality, sediments and the macroinvertebrate community of residential canal estates in south-east queensland, australia: A multivariate analysis. Water Res., 23(9): 1087-1097. https://doi.org/10.1016/0043-1354(89)90152-8

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

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

De'ath, G. and K. Fabricius. 2010. Water quality as a regional driver of coral biodiversity and macroalgae on the great barrier reef. Ecol. Appl., 20(3): 840-850. https://doi.org/10.1890/08-2023.1

Egan, H., 1976. Chemical analysis and water quality. Fresenius J. Anal. Chem., 282(4): 323-329. https://doi.org/10.1007/BF00423583

Elliott, J.M.H., U. H. Humpesch and T.T. Macan. 1988. Larvae of the british ephemeroptera: A key with ecological notes. 49: 145.

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

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

Hall, D.J., W.E. Cooper and E.E.Werner. 1970. An experimental approach to the production dynamics and structure of freshwater animal communities. Limnol. Oceanogr., 15(6): 839-928. https://doi.org/10.4319/lo.1970.15.6.0839

Hawkes, H.A., 1998. Origin and development of the biological monitoring working party score system. Water Res., 32(3): 964-968. https://doi.org/10.1016/S0043-1354(97)00275-3

Hanson, J.M., 1990. Macroinvertebrate size-distributions of two contrasting freshwater macrophyte communities. Freshw. Biol., 24(3): 481-491. https://doi.org/10.1111/j.1365-2427.1990.tb00726.x

Hanson, J.M., E.E. Prepas and W.C. Mackay. 1989. Size distribution of macro-invertebrates communities in fresh water lake. Can. J. Fish. Aquat. Sci., 46: 1510-1519. https://doi.org/10.1139/f89-193

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., M. Weatherhead, C. Stanger and E.Graynoth. 1998. Macroinvertebrate distribution in the littoral zone of lake coleridge, south island, new zealand-effects of habitat stability, wind exposure, and macrophytes. N. Z. J. Mar. Freshw. Res., 32(2): 287-305. https://doi.org/10.1080/00288330.1998.9516826

Kirsty, L.N., C.R. Allen, C. Barichievy, M. Nystrom, S. Sundstrom and N.A.J. Grahan. 2014. Habitat structure and body size distributions: across ecosystem comparison taxa with determinate and indeterminate growth. Oikos. 123(8): 971-983. https://doi.org/10.1111/oik.01314

Leaper, R., D. Raffaelli., C. Emes and B. Manly. 2001. Constraints on body-size distributions: An experimental test of the habitat architecture hypothesis. J. Anim. Ecol., 70(2): 248-259. https://doi.org/10.1111/j.1365-2656.2001.00493.x

Mason, C.F., 2002. Biology of fresh water pollution. Pearson Educated Limited.

McAbendroth, L., P.M. Ramsay, A. Foggo, S.D. Rundle and D.T. Bilton. 2005. Does macrophyte fractal complexity drive invertebrate diversity, biomass and body size distributions? Oikos, 111(2): 279-290. https://doi.org/10.1111/j.0030-1299.2005.13804.x

Morin, A. and D. Nadon. 1991. Size distribution of epilithic lotic invertebrates and implications for community metabolism. J. North Am. Benthol. Soc., 10(3): 300-308. https://doi.org/10.2307/1467603

Ochieng, H., O. Robinson and O. James. 2020. Comparison of temperate and tropical versions of Biological Monitoring Working Party (BMWP) index for assessing water quality of River Aturukuku in Eastern Uganda.

Glob. Ecol. Conserv., 23: 1183. https://doi.org/10.1016/j.gecco.2020.e01183

O'Sullivan, E. and G.R. Rassel. (Eds) 1995. Research methods for public administrators. New York: Longman.

Rasmussen, J., 1993. Patterns in the size structure of littoral zone macroinvertebrates communites. Can. J. Fish. Aquat. Sci., 50: 2192-2207. https://doi.org/10.1139/f93-246

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.

Roche, K.F., E.P. Queiroz, K.O. Righi and G.M. de Souza. 2010. Use of the BMWP and ASPT indexes for monitoring environmental quality in a neotropical stream. Acta Limnol. Brasil., 22(1): 105-108.

Smock, L.A. 1980. Relationships between body size and biomass of aquatic insects. Freshw. Biol., 10(4): 375-383. https://doi.org/10.1111/j.1365-2427.1980.tb01211.x

Roche, K.F., P.Q. Ednilson, O.R. Karina and M. Gláucio. 2010. Use of the BMWP and ASPT indexes for monitoring environmental quality in a neotropical stream. Acta Limnol. Brasil., 22(1): 105-108. https://doi.org/10.4322/actalb.02201010

Stead, T.K., J.M. Schmid-Araya and A.G. Hildrew. 2003. All creatures great and small: Patterns in the stream benthos across a wide range of metazoan body size. Freshw. Biol., 48(3): 532-547. https://doi.org/10.1046/j.1365-2427.2003.01025.x

Strayer, D., 1986. The size structure of a lacustrine zoobenthic community. Oecologia, 69(4): 513-516. https://doi.org/10.1007/BF00410356

Strayer, D.L., 1991. Perspectives on the size structure of lacustrine zoobenthos, its causes, and its consequences. J. North Am. Benthol. Soc., 10(2): 210-221. https://doi.org/10.2307/1467579

Towers, D.J., I.M. Henderson and C.J. Veltman. 1994. Predicting dry weight of new zealand aquatic macroinvertebrates from linear dimensions. N. Z. J. Mar. Freshw. Res., 28(2): 159-166. https://doi.org/10.1080/00288330.1994.9516604

Tamiru, S.M., 2019. Macroinvertebrates as indicators of the water quality of River Shinta, Gondar, Ethiopia. Sustain. Water Resour. Manag., 5: 1227–1241. https://doi.org/10.1007/s40899-018-0297-6

Tsyuyuko, Y., C.L. Piran, W.M. Spencer and D. Raffaelli. 2012. Patterns and processes in abundance-body size relationship for marine bethic invertebrates. Br. Ecol. Soc., 81(2): 463-471. https://doi.org/10.1111/j.1365-2656.2011.01921.x

Vicente, J., M.F. Colmenarejo, E. Sánchez, A. Rubio, M.G. García, R. Borja and A.M. Jimenez. 2009. Evaluation of the water quality in the guadarrama river at the section of las rozas-madrid, spain. Water Environ. J., 25(1): 55-66.

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.