Macroinvertebrates as Bioindicators of Water Pollution

Suheel Dar* and Bashir Ahmad Ganai

Department of Environmental Science, University of Kashmir, Srinagar-190006, J & K, India
Corresponding author: suhailshafi40@gmail.com

Abstract
This review addresses the intersection of water quality policy and benthic macroinvertebrates. Macroinvertebrates, as biological indicators of stream water quality, can be utilized to identify impaired waters, determine aquatic life stressors, set pollutant load reductions, and indicate improvement.
Keywords: Macroinvertebrates, water quality, bioindicator.

Introduction

Macroinvertebrates are almost present everywhere and are a diverse group of long lived species that react strongly and often predictably to anthropogenic activities in aquatic ecosystem. The abundance of benthic fauna greatly depends on physical and chemical properties of the substratum. In aquatic ecosystems Macroinvertebrates act as a barometer of overall biodiversity. Macrozoobenthic invertebrates form the basis of the trophic level and any negative effects caused by pollution in the community structure can in turn affect trophic relationships. They are an important and integral part of any aquatic ecosystem. These can include those that feed on them directly or indirectly such as fish and bird populations, respectively. In addition, they have the ability to clean rivers as aquatic invertebrates utilize the organic and detritus matter. In streams and rivers Macrozoobenthic invertebrate populations can assist in the assessment of the overall health of the stream (Carlisle *et al*., 2007).

Macroinvertebrates as bioindicators

Accordind to Pearson *et al.* (1986) long term changes in the benthic community of two areas Loch Linn and Loch Eil, Scotland. They were of the view that over a period of twenty years the variations in population of benthic community were related to and based upon changing organic inputs which in turn determined the carrying capacity of sedimentary benthos, while as climatic fluctuations like long term temperature changes was responsible for the species composition. A low species diversity index was observed at thermal effluent site due to deteriorated water quality at that site and hence, the community structure at the effluent site was reported to be under stress (Singh, 1988). The population dynamics of the oligochaetes was influenced by the nature of the sediment of the lake as dominant oligochaetes of Dal lake, Kashmir includes *Limnodrilus hoffmeister*, *Tubifex tubifex*, *Branchura sowerbyii*, *Aelosoma* sp. and *Nais* sp. which thrive in sediments rich in organic nutrients (Mir and Yousuf, 2003). In Lake Uluabat, Bursa, Turkey, among 24 taxa of benthic macroinverebrates Insecta and Oligochaeta were the most abundant groups, dominated by species characteristic to nutrient rich waters, including *Pristina aequiseta*, *Nais communis*, *Tubifex tubifex*, *Limnodrilus hoffmeisteri*, *Potamothrix hammoniensis* and

Tanypus punctipennis. Most of the variance (63.5%) in relationships between species and environmental variables as explained by the first two axes of a canonical correspondence analysis (CCA) and placed most Oligochaeta and Chironomidae near the vectors of high nutrients and chlorophyll-a concentrations, while the sensitive Crustacea and some Oligochaeta (Lumbricidae) species on sectors of the plot with the smallest weight of those variables (Celik *et al.*, 2010). In Kishore Sagar the macro-benthic community is dominated by *Chironmus* larvae population. *Chironomus* larvae have also been used as pollution indicators by number of workers (Gaufin, 1957) Thus, the abundance of chironomids in the benthic population is due to impact of altered nature of substrate due to organic pollution. The presence of *Tubifex* sp., *Limnodrilus* sp. and *Limnaea accuminata* in lake also corroborates with the work of Mason (1981). Summer peak abundance of benthic invertebrates was recorded Answar and Siddiqui (1988). It is inferred from the present investigation,that littoral benthic macro-invertebrate community can be useful in evaluating the localized effects of organic enrichment in the lentic systems whereas profundal benthos indicate the water quality as a whole (Mastrantuono, 1986). Benthic macro invertebrates are best indicators for Bio-assessment. The distribution, population density and diversity of the macro benthic community is directly affected by the abiotic environment of the water body. Benthic fauna are especially of great significance for fisheries that they themselves act as food of bottom feeder fishes (Walker *et. al.*, 1991). Any environmental changes in lakes, for example in nutrient concentrations, would be reflected by changes in the structure of the benthic invertebrate community (Carvalho *et al.*, 2002). This means that benthic invertebrates may potentially indicate eutrophication, as planktonic communities, but in addition several other modes of lake degradation. Recent extensive reviews of the current state of the art of ecological water quality assessment systems in Europe have revealed that, while practical assessment tools using macroinvertebrate parameters are already in use to assess the ecological quality of rivers, in many European countries there are currently no working macroinvertebrate assessment systems for lakes (Cardoso *et al.*, 2005). It was suggested by Warwick (1986) that the distribution of individual numbers among species behaved differently from the distribution of biomass among species when influenced by pollution induced disturbance. Also it was reported that combined k-dominance plots for species biomass and number took three possible forms representing unpolluted, moderately polluted and grossly polluted conditions. Pearson and Rosenberg (1978) suggessted that only few species can survive and tolerate in the oxygen deficiency condition by increasing their biomass, such as the capitellid and spionid polychaetes. The increasing of organic materials, correlating with high sulfide and low oxygen, in sediment on transitory zone caused the larger species and deeper burrowing forms gradually eliminated and replaced by greater numbers of lamellibranch suspension and surface deposit feeders, e.g. *Thyasira* and *Corbula*, and holothurian and annelids (Pearson and Rosenberg 1978). The gradients in organic content showed relationship with the polychaete assemblages and opportunistic polychaete species in Kung Krabaen Bay (Chatananthawej, 2001). Two distinctive polychaete assemblages. *Prionospio (Minuspio) japonica*, *Mediomastus* sp.A and *Glycinde* sp.A were polychaetes related with high organic content while *Lumbrineris* sp.B, *Mediomastus* sp.A and *Sigambra* cf. *tentaculata* were related with low organic content sediment in the bay. Chatananthawej (2001) also proposed that a spionid polychaete, *Prionospio (Minuspio) japonica*, could be used as indicator species for moderated organic enrichment area.

The dipterans in the river Jhelum included *Chironomus sp.* and *Diamessa sp.* Presence of bioindicators and *Chironomous sp.* indicates the effect of pollution. The numerical abundance of *Chironomous sp.* throughout the year indicates the pollution status of the river as chironomids are the common inhabitants of polluted waters, water rich in nutrients and water poor in oxygen (Callisto, 2005). The abundance of benthic fauna is greatly related to physical and chemical properties of the substratum. Benthic macro-invertebrates can be used as a barometer of overall biodiversity in aquatic ecosystems (Chatzinikolaou *et al.*, 2006). According to Thomas, 1972 Bio-indicators are useful to assess the functional efficiency of water treatment plants which is one of the important practical considerations. With many new analysis systems there are many different ways to relate benthic macro-invertebrate community structure to water quality. The benthic macroinvertebrate community has been used as an indicator of water quality (Hellawell, 1986). It has been reported that the input of increasing load of pollutants and toxic substance into the surface waters to cause serious disturbances in the aquatic ecosystems. However, this depends on the nature and quantity of pollutants. Usually various physicochemical methods are used to detect the effect of pollution on the water quality changes. Such alterations in water quality are also very well reflected in the structure and composition of biotic community as shown by occurrence, diversity and abundance pattern of species (Kumar *et al.*, 2006). Aquatic macroinvertebrates species have been used by scientists as bioindicators of a river's health. Some are highly sensitive to change while others are heartier and can live under more stressful conditions, such as areas with very low dissolved oxygen levels (Smith, 2007). Aquatic macroinvertebrates live in the marsh grasses and detritus along the edge of rivers. These grasses provide both the shelter and the nutrients needed by these organisms to survive (Parker, 1989). Another thing the macroinvertebrates need to survive is oxygen, generally produced by the plants they live in.

Macroinvertebrates are by far the most commonly group used in the assessment of water quality (Rosenberg and Resh, 1993). Oligochaeta, especially the Tubificidae family, because its capacity of increase in number with increasing organic matter, replacing other benthic macroinvertebrates less tolerant for this condition (Schenková and Helešic, 2006), have been universally applied on bioassessment assays, as bioindicators to reflect the organic pollution in rivers and streams (Lin and Yo, 2008).

Biotic indices that use Oligochaeta as a biological indicator of stream conditions have long been used to evaluate the level of aquatic pollution. Howmiller and Beeton (1971), for instance, consider the high abundance of Oligochaeta as an indication of organic enrichment, while Lafont (1984) on the other hand uses the relative abundance of Tubificidae in a community of Oligochaeta to identify organic enrichment. Moreover, the Howmiller and Scott's index (1977) provides detailed information on the quality of an aquatic habitat because it relies not only on the full identification of all constituent species but also on the knowledge of the ecological demands of a fair number of the most abundant species in the environment.

Studies have shown that species of Oligochaeta Tubificidae are generally tolerant to organic pollutants (Alves *et al.*, 2008). It is possible that what can account for the frequently massive presence of Tubificidae in polluted streams are not only their tolerance to considerable low levels of dissolved oxygen (Aston, 1973) but also a relative decrease in both the rates of competition and predation in such environment (Brinkhurst and Jamieson, 1971). Any environmental changes in lakes, for example in nutrient concentrations, would be reflected by changes in the structure of the benthic invertebrate community (Carvalho *et al.*, 2002). Macroinvertebrates are a good bioindicator of these changes of water quality and are used frequently in measuring the anthropogenic effects on aquatic environments (Couceiro, 2012). A bioindicator is a species or group that "readily reflects" the state of its environment and can represent the impacts of environmental change on an ecosystem (Hodkinson, 2006). The small size, quick reproduction rate, and high density of macroinvertebrates lead it to become excellent bioindicator. Aquatic macroinvertebrates, in specific, respond quickly to changes in the environment and the abundance of between species of different tolerant levels show the degradation of their environment. In addition, being the second largest group of aquatic organisms, they are responsible for the maintenance of their ecosystems and are important in nutrient cycles and as an energy source for other organisms (Couceiro, 2012). The macroinvertebrates are very sensitive to changes in several abiotic parameters of the aquatic habitats. In Asan wetland, the density of macroinvertebrates was positively correlated with the amount of dissolved oxygen (y = 163.01x + 775.91 R2 = 0.8838). The amount of dissolved oxygen was higher at two sites, which may be attributed to the presence of high amounts of macrophytes in the littoral zone of the wetland. A negative relationship was found between the density of macroinvertebrates and turbidity (y = _2.5721x + 707.34 R2 = 0.87) and total dissolved solids (y = _4.172x + 970.02 R2 = 0.7227). However, a positive relation was found between the density of macroinvertebrates and transparency (y = 513.07x + 30.115 R2 = 0.6855). A negative relationship (y = _231.87x + 1078.8 R2 = 0.8154) was also found between the density of macroinvertebrates and hydromedian depth. Density of macroinvertebrates showed a negative relation (y = _30.111x + 1027.7 R2 = 0.6099) with water temperature. Density of macroinvertebrates was found to be maximum (533 indm_2) at one site, which may be due to presence of riparian vegetation, suitable substrates and minimum water level fluctuations. The riparian vegetation may provide them protection from predators and suitable environment for the growth of periphytic algae, which is an important food source for many macroinvertebrates. Most of the invertebrates utilize plants as a direct food source, sites for oviposition and source of respiratory oxygen (Krull, 1970).

Most importantly, macrozoobenthos have been extensively used for the assessment of the ecological integrity and biomonitoring of aquatic habitats (Acharyya *et al.,* 2001). This is because they manifest a distinct response to changes in the aquatic environment, thus serving as promising indicators of hydrologic stress and aquatic ecosystem health in general (Nazarova *et al.*, 2004). In addition, the sedentary nature of macrozoobenthos, together with their ubiquitous distribution and lifecycles of measurable duration allow for both long-term and short-term analyses, and they are easy to identify with already established diversity and monitoring indices (Rosenberg and Resh 1993).

Environmental factors affecting macroinvertebrates

Invertebrate communities change in response to changes in physicochemical factors and available habitats. The biotic structure and water quality of streams and rivers reflect an integration of the physical, chemical and anthropogenic processes occurring in a catchment area, leading to the concept of ecological integrity. Human induced hydrological changes, physical disturbances (habitat alteration, urban land use) and point and nonpoint sources of pollution (chemical contamination, surface runoff, intensive agriculture) are examples of processes responsible for a broad-scale deterioration of lotic ecosystems. The river receives untreated effluents from many urban settlements and industrial discharges in both countries (Chatzinikolaou *et al.,* 2006). Those parameters are also defined of prime importance for DMills (1972), who stated that the chief environmental factors affecting distribution of aquatic animals in streams are: The chemical nature of the water, this may affect the distribution of aquatic organisms in a number of ways. The concentration of dissolved oxygen is important. Oxygen is not very soluble in water and its solubility depends on the temperature. The calcium content of water is also important for species such as the freshwater shrimp, many snails and mussels, which are abundant in hard water. The physical nature of the water, water movement, and temperature. The surface velocity of water has been shown to have an effect on caddis larvae and it was found that the number of Trichoptera larvae along a current gradient became progressively less numerous with increasing distance from a source. benthos cannot move around as much, so they are less able to escape the effects of sediment and other pollutants that diminish water quality. Therefore, benthos can give reliable information on stream and lake water quality. Their long life cycles allow studies conducted by aquatic ecologists to determine any decline in environmental quality. Macro invertebrates constitute a heterogeneous assemblage of animal phyla and consequently it is probable that some members will respond to whatever stresses are placed upon them (Hellawell, 1986). Temperature has been implicated as a mechanism influencing spatial and temporal isolation (Ward, 1992), and as one of several primary factors influencing life history patterns of aquatic insects (Sweeney, 1984). In earlier researches, Grenier (1949), showed that population of Simuliidae differs in genera to their occurrence in altitude, thus in temperature. Various aspects of water chemistry, e.g., acidity, dissolved oxygen; water hardness, etc. (Foster, 1991) also influence the distribution of freshwater taxa, although ascertaining the effects of selected chemical factors can be difficult (Ward, 1992). Newlon and Rabe (1977) stated that the two most important factors affecting macroinvertebrates are substrate and suspended sediment. They found that there are four to five physical and chemical factors that have significant influence over biomass and diversity of macroinvertebrates. These factors include substrate, suspended sediment, gradient, water temperature, and stream order and width. Minshall (1984) supports these findings and provides a literature review of insect-substratum relationships. Peeters *et al.* (2000) have defined in their study that current velocity and substratum are the two main physical factors affecting distribution of lotic macroinvertebrates. Earlier researches tend to demonstrate that substratum/organism relationship was not well understood due to a lack of research on that subject (Hynes, 1970). According to Jumppanen (1976) also the first signs of eutrophication and pollution in some Finnish lakes are usually seen in the benthic fauna as the suspended wastes immediately sink to the bottom to decompose and thus causing a change in the benthic organisms. Thus, certain species of sponges, for exam-ple, respond to various types of poisonous pollutants even in very mild cases, while others (tubificids, sludge worms, maggots and chironomids) can tolerate even the most gross organic pollution and high levels of toxic pollution. Hence, the species analysis of benthic commu-nity enables the determination of trophic type of lakes and is, therefore, an important criterion in the ecological classification of lakes (Bazzanti, 1975). As the emergence of species like *Tubifex* sp. and *Chironomus* sp. in Nilnag lake indicated the eutrophic status of the lake (Yaqoob *et al.*, 2007). Salinity is the major factor determining the composition, diversity and abundance of macrozoobenthic fauna in coastal lagoons in Ghana (Gordon, 2004 and Lamptey, 2008), with high macroinvertebrate diversity and abundance occurring at low salinities, and low diversity and abundance occurring at high salinities. It is therefore possible that the difference in salinity levels of the two lagoons is responsible for the difference in the structure and composition of their macrozoobenthos communities, with the community of Amansuri mostly being insect larvae while that of Domini largely worms (annelids) and marine crustaceans. This explains the very low similarity value (*Cs*= 0.3) which indicates that the two communities are nearly dissimilar. Pertinently, some of the insects found in Amansuri, especially the Chironomid larvae, are reportedly intolerant to salinities beyond 6 ‰ (Bervoets *et al*., 1996 and Berezina, 2004), hence the freshwater nature of the lagoon is favourable for their survival and development.

pH being one of the most important water quality parameters has been found to have profound effects on the ecology of macroinvertebrates in aquatic systems. Although, benthic macroinvertebrate sensitivities to pH vary (Yuan, 2004), values below 5.0 and greater than 9.0 are considered harmful. Low pH values are associated with lower diversity of benthic macroinvertebrates (Thomsen and Friberg, 2002), and cause decreased emergence rates in them (Hall *et al.*, 1980). In macroinvertebrates, low pH has also been associated with egg failure (Willoughby and Mappin, 1988) and physiological problems because it is difficult for benthic macroinvertebrates to regulate ions within their bodies and to absorb the calcium needed for exoskeletons (Hall *et al.*, 1980). A decrease in the pH of stream water can trigger the release of heavy metals, which are toxic to benthic macroinvertebrates (Peiffer *et al.*, 1997).

Amphipods, isopods, crayfish, snails, and bivalves are more common in hard than in soft waters, as has been reported by several workers (Allan, 1995).

Conclusion

Aquatic macro-invertebrates play significant role in responding to a variety of environmental conditions of rivers, streams and Lakes, therefore may be used as bio-indicators for water quality assessment. Macroinvertebrates occupy an important position in the lake ecosystem serving as a link between primary producers and higher trophic levels. They also play an important role in the decomposer food chain which in turn affects the cycling of minerals. The occurrence, composition and distribution of macroinvertebrates in lakes and wetlands is governed by numerous environmental factors that affect the structure of macrozoobenthic community and their distribution pattern should be considered while evaluating the ecological status of Lakes.

References

Acharyya S; WJ Mitsch. **2001**; Macroinvertebrate diversity and its ecological implications in two created wetland ecosystems. p 65-76. In: ***The Olentangy River Wetland Research Park at The Ohio State University Annual Report 2000*** (W.J. Mitsch and L. Zhang, eds.), The Ohio State University, Columbus, USA

Allan JD 1995. Stream Ecology: Structure and Function of Running Waters. Chapman and Hall, New York, 388 pp.

Alves, RG., Marchese, MR. and Martins, RT. 2008,Oligochaeta (Annelida, Clitellata) of lotic environments at Parque Estadual Intervales (São Paulo, Brazil). ***Biota Neotrop***., **18 (1)**: 21-24.

Anwar, S. and Siddiqui, M.S. 1988. On the distribution and dynamics of macroinvertebrate fauna of the river Kali in Northern India. ***Journal of Environmental Biology***. **9:** 333-344.

Aston, RI. Tubificids and water quality: a review. ***Environ. Pollut***., 1973, vol. 5, no. 1, p. 1-10.

Bazzanti M 1975. Chironomidi (Diptera) dei Sedimenti del lago di Martignano (Lazio). ***Bull. Pesca Piscic. Idrobiol.*** **30:** 139-142.

Berezina, N. A. **2004.** ***Russian Journal of Ecology***, , **34**(4), 261-266.

Brinkhurst, R. O. and Jamieson, B. G.M. 1971.Aquatic Oligochaeta of the World. 1a ed. Toronto: University of Toronto Press, 860 pp.

Callisto, M., Goulart. M., Barbosa,,F. A. R. and Rocha. O. 2005. Biodiversity assessment of benthic macroinvertebrates along a reservoir cascade in the lower sao Francisco River (Northeastern Brazil). ***Braz. J. Biol***. **65 (2):** 1-6.

Cardoso, A. C., Solimini A. G, Premazzi G., Birck S., Hale P., Rafael T., Serrano M. L. 2005. Report on harmonisation of freshwater biological methods. European Commission Joint Research Centre, ***Report EUR 21769 EN,*** p. 60-80, Ispra.

Carlisle, D. M., Meador, M. R., Moulton, S. R. and Ruhl, P. M. 2007: Estimation and application of indicator values for common macroinvertebrate. ***Ecol. Indicators,*** **7:** 22-33.

Carvalho, L., H. Bennion, A. Darwell, I. Gunn, A. Lyle, D. Monteith, and M. Wade, 2002. Physicochemical Conditions for Supporting Different Levels of Biological Quality for the Water Framework Directive for Freshwaters. ***Report to the Environment Agency***, UK. p. 16-18

Celik K, Akbulut N, Akbulut A, Ozatli D. 2010. Macrozoobenthos of Lake Uluabat, Turkey, related to some physical and chemical parameters. Panam. ***J. Aquat. Sci***. 5(4): 520-529.

Chatananthawej, B. 2001. Ecological Studies on Benthic Polychaetes with Respect to Organic Enrichment Condition in Kung Krabaen Bay, Thailand. Ph.D. Thesis, Chulalongkorn University, Bangkok.

Chatzinikolaou, Y., Dakos V., And Lazaridou, D. 2006: Longitudinal impacts of anthropogenic
pressures on benthic macroinvertebrate assemblages in a large transboundary Mediterranean river during the low flow period. ***Acta hydrochim. Hydrobiol***. **34:** 453- 463.

Couceiro SRM, Hamada N, Forsberg BR, Pimentel TP et al 2012 A macroinvertebrate multimetric index to evaluate the biological condition of streams in the Central Amazon region of Brazil. ***Ecol Ind.*** **18:**118–125

Gaufin, A. R. 1957. The effects of pollution of a mid western stream, ***Ohio. J. Sic***. **58(4):**197-208.

Gordon C. **1987**. Fisheries in small water bodies: the case of the Amansuri Wetlands, Ghana. p. 70-174, In: ***Symposium on the development and management of fisheries in small water bodies.,*** 7th-8th Dec. **1987,** Accra, Ghana. (CIFA/87/SYMP./8) FAO Fisheries Report No. 425.

Grenier, P. 1949. Contributio a l'etude biologique des Simuliidae de France. ***Physiologia comp. Oecol.*** **111:** 165-333

Hall RJ, Likens GE, Fiance SB, Hendrey GR 1980. Experimental acidification of a stream in the Hubbard Brook experimental forest, New Hampshire. ***Ecology,*** **61:**976-989

Hellawell, J.M. 1986. Biological indicators of freshwater pollution and environmental management. ***Elsevier***, London, 518 pp.

Hodkinson I. D. and J. K. Jackson, 2005. Terrestrial and aquatic invertebrates as bioindicators for environmental monitoring, with particular reference to mountain ecosystems," **Environmental Management, 35 (5):** 649–666.

Howmiller, RP. and Beeton, AM. 1971. Biological evaluation of environmental quality, Green Bay, Lake Michigan***. J. Water Pollut.*** **42 (3):** 123-133.

Hynes H. B. N. 1970. ***The Ecology of Running Waters***. Liverpool University Press, pp. 555

Jumppanen K 1976. Effects of waste waters on a lake ecosystem. ***Ann. Zool. Fennici***. **13:** 85-138.

Krull, J.N., 1970. Aquatic plant-macroinvertebrate associations and waterfowl. ***J. Wildl. Manage***., 34: 707-718.

Kumar, A., Qureshi, T. A. and Alka, P. 2006. Biodiversity assessment of macroinvertebretes in Ranjit Sagar reservoir, Jammu, J&K, India. ***J. Aqua. Biol***. 21(2):39-44.

Lafont, M. 1984. Oligochaete communities as biological descriptors of pollution in the fine sediments of rivers. ***Hydrobiologia***, **115 (1):** 127-129.

Lamptey E; AK Armah. **2008**. ***Estuaries and Coasts***,**31:** 1006-1019.

Lin, KJ. and Yo, SP. 2008. The effect of organic pollution on the abundance and distribution of aquatic oligochaetes in an urban water basin, Taiwan. ***Hydrobiologia***, **596 (1):** 213-223.

Mason, C.F. 1981. ***Biology of Freshwater Pollution***. Longman, London. pp.209

Mastrantuono, I.1986. Littoral sand zoobenthos and its relation to organic pollution in Lake Nemi9Central Itlay). ***Hydrobiological Bulletin.*** 19(2): 171-178.

Mills, D. H. 1972. ***An Introduction to Freshwater ecology***. Cox & Wyman Ltd.

Minshall, G.W. 1984. Aquatic insect-substratum relationships. p. 358-400. In: ***The Ecology of Aquatic Insects*** **(**V. H. Resh and D. M. Rosenberg. eds). Praeger Publisher. New York.

Mir, M.F., Yousuf, A. R. 2003. Oligochaete community of Dal lake, Kashmir. ***Orient. Sci***. 8: 83-87.

Nazarova L; B Semenov; VF Sabirov; RM Efimov; L Yu. ***Water Resources***, **2004**, **31**(3), 316–322.

Newlon, T.A. and Rabe, F.W. 1977. Comparison of macroinvertebrate samplers and the relationship of environmental factors to biomass and diversity variability in a small watershed. Research

Technical Completion Report, p. 33-46, project A-049-IDA; Idaho Water Resources Research Institute, University of Idaho, Moscow.

Parker MS. 1989. Effect of substrate composition on detritus accumulation and macro invertebrate distribution in a southern Nevada desert stream. SW Nat. 34:181–187.

Pearson, T.H. and Rosenberg, R. 1978. Macrobenthic Succession in Relation to Organic Enrichment and Pollution of the Marine Environment. ***Oceanogr. Mar. Biol. Ann.*** **Rev.** 16: 229 - 311.

Pearson TH, Duncan G, Nuttal J. 1986. Long term changes in benthic communities of Loch Linnhe and Loch Eil, Scotland. ***Hydrobiologia.*** 142: 113-119.

Peeters E. T. H. M., S. Batten, J. A. J. Beijer, and J. J. P. Gardeniers. 2000. The combined effects of current velocity and substratum particle size on the performance of Gammarus pulex, an experimental approach. Aquatic Ecology and Water Quality Management Group, Environmental Sciences, Wageningen University, The Netherlands. pp. 327-329.

Rosenberg DM, Resh VH 1993. Introduction to Freshwater Biomonitoring and Benthic Macroinvertebrates. 1-9. In: ***Freshwater Biomonitoring and Benthic Macro-Invertebrates. (***Rosenberg, D. M. and V. H. Resh. eds.). Chapman and Hall, New York pp:

Schenková, J. and Helešic, J. 2006. Habitat preferences of aquatic Oligochaeta (Annelida) in the Rokttná River, Czech Republic- a small highland stream. ***Hydrobiologia***. 564 (1): 117-126.

Singh S 1988. Evaluation of macrobenthic community through species index in Kanti Oxbow lake, Muzzaffarpur, Bihar. ***J. Int. Fish. Soc. India*** 20(2): 30-34.

Smith , A.J. , R.W. Bode , and G.S. Kleppel. 2007 . A nutrient biotic index (NBI) for use with benthic macroinvertebrate communities . ***Ecological Indicators*** 7: 371 – 386

Thomas, W.A. 1972. Indicators of environmental quality. Plenum Press, New York, pp.240.

Thomsen AG, Friberg N 2002. Growth and emergence of the stonefly Leuctra nigra in coniferous forest streams with contrasting pH. ***Freshwater Biol***. 47: 1159-1172

Walker, I., Henderson, P.A. and Sterry, P. 1991 - On the pattern of biomass transfer in the benthic fauna of an Amazonian black water as evidenced by 32 P level experiment. ***Hydrobiol***., 215: 153-162.

Ward, J.V. 1992. Aquatic Insect Ecology: 1. ***Biology and Habitat. John Wiley and Sons***, New York. pp. 438

Warwick, R.M. 1986. A New Method for Detecting Pollution Effects on Marine Macrobenthic Communities. ***Mar. Biol***. 92: 557 - 562.

Yaqoob KU, Wani SA, Pandit AK 2007. A comparative study of macro-benthic community in Dal and Nilnag lakes of Kashmir Himalaya. ***J. Himalayan Ecol. Sustain. Dev***. 2: 55-60.

Yuan L.L. 2004. Assigning macroinvertebrate tolerance classifications using generalized additive models. ***Freshwater Biol***. 49: 662-677.