CHAPTER 2
LITERATURE REVIEW

The literature survey below have been concentrated on the use of macroinvertebrates in biological monitoring with reference to the requirements of ideal bioindicators, the advantages of the use of macroinvertebrates as bioindicators, research work performed using macroinvertebrates as bioindicators locally as well as globally, and new approaches in biological monitoring. A discussion of macroinvertebrate sampling methods deals with both conventional and artificial substrate samplers, their designs, advantages and disadvantages as well as some research work carried out using these methods. Data analysis section describes indices used in biological monitoring works, their merits and demerits.

2.1 Use of Macroinvertebrates as bioindicator

Approaches to the biological monitoring of fresh water using macroinvertebrates have diversified in recent decades. Three possible responses of a community to environmental change have been identified . These are i) biomass changes, but the same community structure is maintained ii) community structure changes, and biomass may change but the same species are maintained iii) species and community structure change, and biomass may change (Hellawell, 1978, 1979) (Appendix A 1).

Perhaps the most common type of biomonitoring using benthic macroinvertebrates is termed surveillance. It includes surveys done before and after a project is completed, or before and after toxicant is spilled, or to determine if water resource management techniques are working, or whether conservation measures are successful (Hellawell, 1986; Abel, 1989; Rosenberg *et al.*,1993). In addition, biomonitoring is done to ensure compliance, with regulations either to meet immediate and statutory requirement or to control long term water quality (McBride, 1985 as cited in Rosenberg *et al.*, 1993).

Benthic macroinvertebrates are used to achieve the above objectives in a variety of ways, including monitoring changes in genetic composition, bio-accumulation of toxicants, toxicological testing in the laboratory and field, and measurement of changes in population number, community structure or ecosystem functioning (Rosenberg *et al.*, 1993).

Community based methods of biological monitoring are widely used by environmental agencies in many parts of the world, such as Europe and North America, but are conspicuously absent in others, notably in many tropical regions (Thorne, 1993). However as the capacity for chemical monitoring may be limited but the biological sciences are suitably advanced, biological monitoring is often particularly appropriate for tropical developing countries.

The concept of bioindicators of environmental condition originated with work of Kolkwitz and Marsson (1908, 1909), who developed the saprobien system (as cited in Hellawell, 1986). Saprobien system is based on fact that in rivers subject to organic pollution, community downstream of the pollution input show a regular and more or less predictable sequences of changes in the presence and abundance of indicator species. Hellawell (1986) listed the following requirements for an ideal indicator species. i) They are readily identified ii) They may be sampled easily and quantitatively iii) They have a cosmopolitan distribution iv) Their autecological data are available v) They have economic importance as a resource or nuisance or pest vi) They can readily accumulate pollutants vii) They are easily cultured in the laboratory viii) They have low variability, both genetic and in relation to their role in the biological community.

A literature survey on the use of bioindicators made by Hellawell (1986) revealed that algae and macroinvertebrates are the groups of organisms most often recommended for use in assessing water quality, but in practice macroinvertebrates and algae are the most commonly used group.

Benthic macroinvertebrates offer many advantages in biomonitoring over other indicator species. Hellawell (1986) and Abel (1989) listed several advantages. These are i) they are ubiquitous, so that they can be affected by environmental perturbation in many different types of water bodies ii) the large number of species involved offers a spectrum of responses to environmental stresses iii) their sedentary nature allows effective spatial analyses of pollutant or disturbance effects iv) they have long life cycles, which allow the assessment of temporal changes caused by perturbations.

In addition, various technical developments have enabled benthic macroinvertebrates to be used advantageously in biomonitoring programs. These are i) sampling and sample analyses methods are well developed and can be achieved with simple equipment ii) the taxonomy of many groups is well known and keys to identification are available iii) many methods of data analyses have been developed and are widely used in community level biomonitoring iv) the responses of many common species to different type of pollution have been established v) benthic macroinvertebrates are particularly well-suited to experimental approaches to biomonitoring (Hellawell, 1986; Abel, 1989; and Hawkes, 1979; Penny, 1983 as cited in Rosenberg , 1993).

Some difficulties can also be found in the use of benthic macroinvertebrtates in biomonitoring. These include failure to indicate stress, and problems of study design and analysis (Rosenberg *et al.*, 1993). Concerning study design, quantitative sampling is difficult because of the contagious distribution of benthic macroinvertebrates. This requires a high number of samples to achieve desirable precision in estimating population abundance (Hellawell, 1986).

There is ample evidence of the use of benthic macroinvertebrates as bioindicators for the assessment of water quality all over the world. Pinder *et al.* (1987)

carried out comprehensive study to evaluate water quality in River Frome, using bioindicators. To perform this study they used values of the Chandler score, NWC (National Water Council) score and average NWC score per taxon and results showed low values for all above mentioned indices indicating lower water quality at study sites. Dudgeon (1984) studied the Lam Tsuen river in Hong Kong, data based on macroinvertebrates distribution and abundance.

Chemical monitoring is well established and popular in Thailand. But problems are encountered in the cost and availability of equipment. As such, researchers using biomonitoring methods to assess water quality are becoming more popular. For example, comprehensive biomonitoring researches have been carried out in Northern Thailand. Sannarm (1993) carried out a research to investigate water quality and macroinvertebrate communities in the Mae Kwung river, in the vicinity of the Northern region industrial estate, in Chiang Mai. She found there was no evidence that the industrial estate created serious water pollution, but there were interesting differences between the fauna of various microhabitats, and seasonal patterns.

Research carried out to investigate water loss at the old Mae Ping dam and water quality of the Mae Ping river using biomonitoring techniques, by Thaweeburus (1994) revealed an inverse relationship between taxa richness and conductivity. Further, she found that the effect of sewage at the outlet of the Mae Kha canal was greater than the damming effect on the water quality of the river. She also pointed out that the different sampling techniques and seasons could effect the number of taxa of macroinvertebrate found.

Rajchapakdee (1992) found a strong relationship between altitude and the types of macroinvertebrates present in two streams on Doi Suthep in Northern Thailand. She pointed out a relationship between water current and number of species. Research carried out by Tuyor (1993) to investigate the impact of highland agriculture on stream macroinvertebrates in Ban Nong Hoi and Doi Chiang Kian, two different types of streams in Northern Thailand, revealed the combined effect of pesticides and the loss of microhabitat due to siltation as the major factors which affected the macroinvertebrate communities in the study area. Research carried out to assess water quality in different lotic system in Chiang Mai province using quantitative surveys and habitat assessment revealed a relationship between the physico-chemical properties of the water and the habitat quality (Watchawuong, 1996).

Rapid assessment approaches are popular due to their efficiency and cost effectiveness. A lower budget and saving of time is achieved by reduced sampling and more efficient data analysis. Rapid bioassessment programs are designed to screen large regions, pinpointing trouble spots worthy of more detailed attention (Resh *et al.*, 1995). Rapid bioassessment based on quality classification of streams always comparisons to be made between a reference area and areas of concern.

The Rapid Bioassessment Protocol (RBP) was developed by the US Environmental Protection Agencies (EPA) to provide basic aquatic life data for planning and management purposes. This protocol consists of three macroinvertebrate and two fish protocols. Benthic macroinvertebrates Rapid Bioassessment Protocol I (RBP I) and fish Rapid Bioassessment Protocol IV (RBP IV) are cost-effective, screening procedures that provide some initial data which can support further investigation. RBP (II) can be used to prioritize sites for more intensive evaluation or can be used instead of RBP (I) as a screening techniques. It consider only family level taxonomic identification and therefore involves little additional time and effort. RBP (II) provides more intense assessment than RBP (I) and can detect sites of intermediate impairment. Benthic Rapid Bioassessment Protocol III (RBP III) and fish RBP (V) can help set priorities for more intensive evaluation. Benthic RBP (III) and fish RBP (V) are progressively more rigorous and provide more conformatory data but also require more resources (Plafkin *et al.*, 1989).

Chessman (1995) carried out rapid bioassesssment procedure based on macroinvertebrates to investigate the degree of pollution in rivers and streams in Eastern Australia. The procedure involved obtaining standardized collections of 100 animals from six different habitat types. A biotic index, the Stream Invertebrates Grade Number Average Level (SIGNAL), which is calculated by modification of the Biological Monitoring Working Party (BMWP) score was used to assess water quality.

Growns *et al.* (1995) carried out a comprehensive study on rapid bioassessment of river using macroinvertebrates based on habitat specific sampling in the Nepean river and rivers in the Blue mountain area, New South Wales in Australia. The Nepean data showed that the sample size of 100 animals used in the rapid procedure was sufficient to reveal natural distribution patterns in the communities, and that SIGNAL was essentially independent of these patterns. However, the Blue Mountain data showed that water pollution had a greater effect on macroinvertebrate communities than the physical habitat, and SIGNAL distinguished sites with differing levels of pollution.

Suwanrat (1996) conducted research in order to determine the applicability of RBP (II) in Thailand. The results revealed that the RBP (II) could possibly be used in Thailand. However, some improvements or adjustments are needed before using this protocol, as some indices used, such as EPT index and percent contribution of dominant family, were less effective than the others at measuring water quality. Additionally she recommended the use of artificial substrate samplers or dredge samplers as suitable sampling techniques because she met problems while using kick nets and pond nets for macroinvertebrate sampling.

Different levels of taxonomic resolution i.e. species level, family level and order level, strongly affect the data interpretation (Wright *et al.*, 1995). Research performed to measure the impact of sewage effluent on the macroinvertebrates community in upland streams in NSW , Australia, revealed that binary data (presence or absence) at the order level are likely to be sufficient for detecting gross disturbances, whilst moderate disturbances can detected by quantitative methods or at least family level identification. Further they pointed out that quantitative methods and species level identification are most appropriate for assessing subtle differences in macroinvertebrate communities.

2.2 Macroinvertebrates Sampling Methods

The validity, accuracy and precision of any biomonitoring program depends crucially upon the sampling methods and strategy adopted at its outset. Depending on the objectives of the investigation and the type of analysis to which the data are to be subjected, either qualitative or quantitative or both methods are used in biomonitoring programs.

The commonly used sampling methods for collecting macroinvertebrates can be described under three main headings (Hellawell, 1986).

i) methods that extract and separate organisms by disturbing their habitat - e.g. nets, Surber sampler etc..

ii) methods used to collect animals by removing an appropriate part of the habitat together with its associated organisms- e.g. grabs, cores

iii) methods that provide habitat and time for colonizing organisms during a colonization period - e.g. artificial substrate samplers.

Methods 1 and 2 can be categorized as active sampling methods while method 3 is a passive method. The present study deals with all 3 types of sampling methods mentioned above. In addition, sampling methods can be classified as quantitative or qualitative. Surber type sampler, cylinder samplers, grab type samplers, core samplers, and air lift samplers are quantitative sampling methods. Hand nets, shovels, dredges, and drift nets are qualitative sampling methods (Appendix A 2) (American Public Health Association, 1978).

2.2.1 Conventional Methods

The most widely used and simplest conventional macroinvertebrate sampling method is the kick sampler. It is a purely qualitative sampling method, although some operators attempt to introduce a quantitative element by sampling from a known area or for a standard period of time. However, it is difficult in practice to standardize adequately either the area sampled or the duration of a sampling period, and entirely impossible to standardize the actual sampling effort (Abel, 1989). Furse *et al.* (1981) pointed out that the numbers of families and species caught indicate significant difference with respect to sites, operators and the site × operator interaction. Significant inter-operator differences in the number of taxa removed from samples were also shown at the sampling processing stage, but only at the family level.

The Surber sampler is designed to overcome some of these problems associated with operator variability and the standardization of sampling effort and sampling area. Since the Surber sampler can be used only in fairly shallow parts, grabs or dredges are designed for deeper part of the water bodies. These are available in wide variety of designs. Grabs are especially suitable for fine substrates but their efficiency is often low (Abel, 1989).

Using plastic beads as "animals" Elliot and Dreak (1981a) compared the efficiencies of seven commonly-used grab sampler designs. The data revealed that the efficiency of all grab samplers was low when the model particle size exceeded 16 mm, and the grabs in general operated inefficiently, when the animals were buried 3 cm below the surface.

Dredge samplers can be used to sample deep water bodies with coarse substratum. The dredge is dragged along the sampling bed collecting substratum and animals in the collecting net. The relative efficiency of four dredges as qualitative samplers were assessed in field trials by Elliott and Dreak (1981b). The qualitative five samples as a percentage of the total number of taxa caught at each site by four dredges . Values ranged from 76% to 40% for the most efficient and least efficient respectively.

2.2.2 Artificial Substrate Samplers (ASS) as a sampling method

After realizing that all aquatic habitats could not be sampled effectively by conventional methods viz., grabs or nets, researchers opted to use of artificial substrate or colonization samplers for macroinvertebrates (Beak *et al.*, 1973). More recently emphasis has been placed on using artificial substrates to reduce the variability associated with conventional sampling devices (Hellawell, 1978).

An artificial substrate sampler is an item of field equipment that mimics certain features of the aquatic environment into which it is placed. The sampler is colonized by macroinvertebrates at different rates and can be retrieved after an appropriate period of colonization (Beak *et al.*, 1973). Hellawell (1978) predicted that artificial substrate "methods are likely to become increasingly important in the routine surveillance of rivers".

2.3 Types of ASS use for sampling fresh water benthic macroinvertebrates

Artificial substrate samplers (ASS) can be divided into two types. ASS that closely resemble the natural substrate over, on or within which they are placed as Representative Artificial Substrate (RAS), e.g. tray filled with rock or stones collected from stream or river, and samplers that differ from the natural substrate of the habitat in which they are placed, the so-called Standardized Artificial Substrates (SAS), e.g. tempered hardboard multiplate sampler suspended over a strong stream substratum

(Rosenberg *et al.*, 1982). In the present study, the wire mesh cage filled up with stones and the wooden box filled up with grass can be regarded as RAS, and the multi-plate sampler made up with baked clay tiles can be regarded as SAS.

Eight major categories of artificial substrate samplers were identified (Rosenberg *et al.*, 1982).

i. Containers filled with various substrates
ii. Multiplates
iii. Boards, panels and tiles
iv. Bricks and blocks
v. Plastic sheets, polythene and fabric strips
vi. Implanted substrates
vi. Natural organic substrates
viii. Miscellaneous substrates

Category one included, pans and trays, boxes and baskets, filled with rocks or other substrates. Hilsenholf (1969) used a cylindrical sampler made with solid galvanized iron with 3.6 kg of limestone sandwiched between pieces of 16 gauge galvanized hardware cloth inserted inside cylinder. Flannagan *et al.* (1982) used plastic trays filled with granite stones of various sizes, suspended below a 300 μm mesh screen. Minshall and Minshall (1977) used trays with a wood frame and perforated hardwood base covered with nylon mesh. Roux *et al.* (1976) used plastic wire basket reinforced with metal rods and having a lid of metal mesh filled with limestone rocks. Bergersen and Galae (1975) used barbecue baskets filled with 5-7.5 cm pieces of heat treated commercially available coniferous tree bark.

Alternating 7.6 cm and 2.5 cm squares of 0.3 cm thick masonite mounted on a centrally positioned bolt was used as ASS by Hester and Dendy (1962) (as cited in Flannagan, 1982). This is regard as the original Hester-Dendy multiplate sampler and later many authors modified this by changing number of plates, shape of the plate and method of anchoring.

Fullner (1971) used modified Hester-Dendy sampler with 14 circular plates and 24 spacers anchored on a rod to compare efficiency with basket sampler. Arthur and Horning (1969) used five sets of Hester-Dendy samplers mounted on a 1.2 m cross-piece in pollution surveys.

Cover *et al.* (1978) used a modified Parsons-Tatum (1974) model composed of eight circular plates of tempered hard-board instead of square which smooth and rough sides alternated in the sampler. Meier *et al.* (1979) used a modified Hester-Dendy (1962) multi-plate sampler which used four plates instead of eight and which was anchored to the stream bottom by driving the rod threating the plates into the substrate. They determine the rate of colonization by macroinvertebrates in gravel substrate streams. As mentioned in Flannagan (1982) and Konstantinov (1977) used vertically and horizontally placed wooden planks in lakes as ASS, and Gersabeck and Merritt (1979) used white cement tiles and clear plastic tapes, set at various depth and on the bottom as ASS. Hoar and Miller (1972) used concrete and wood blocks in pairs as ASS; Markosova (1980) used granite cubes as ASS by suspending various depths (as cited in Flannagan, 1982).

Many authors used natural organic substrates as ASS. Peterson and Cummins (1974), as cited in Flannagan (1982) used 10 g leaf packs of oven-dried leaves of a single species fastened together with plastic buttoneers and tied with nylon monofilament line to approximately 1 kg brick , positioning the leaves on top facing into the current.

2.4 Advantages and Disadvantages of ASS

Many authors and agencies discussed the advantages and disadvantages of the use of ASS as a macroinvertebrate sampling method. Rosenberg *et al.* (1982) pointed out several advantageous and disadvantageous features.

2.4.1 Advantages

I. Allow collection of data from locations that can not be sampled effectively by other means.

Deep habitats with hard or shifting substrates and large boulders can not be sampled by Ekman grab or core samplers. In addition light weight Ekman grabs can not operate properly in streams or rivers with high flow rates especially in the rainy season. In some situations, ASS can be placed and retrieved during a range of weather and stream conditions (Crowe, 1974; Armitage, 1979). Additionally, ASS can be used in sampling sites where researchers prefer not to use conventional methods, such as highly polluted streams, sewage disposal sites, etc..

II. Permit standardized sampling

This is achieved by eliminating i) subjectivity in choice of microhabitat to be sampled ii) subjectivity in actually taking sample, and iii) confounding effects of differences in habitat. The overall role of the operator is less important in taking ASS than direct sampling methods. The efficiency and precision of direct sampling methods are influenced by the operator.

III. Reduce variability compared with other types of sampling.

Artificial substrate generally provide less sampling variability and increased sampling precision over direct sampling methods. Rosenberg *et al.* (1982) concluded that artificial substrates generally require fewer sampler replicates than direct sampling devices to achieve a given sampling precision. Several authors have found reduced sampling variability for SAS (Weber, 1973; Beak, 1973; Rabeni and Gibbs, 1978) and for RAS (Shaw and Minshall, 1980).

IV. Require less operator skill compared to other methods.

The setting and retrieval of artificial substrates can be done by a less skilled operator. This will lead to reduced labour costs by hiring non biologists. However, more complex artificial substrates especially those requiring field sorting and identification from the sampler may cause difficulties for nonspecialists.

V. Convenient to use

The following aspects describe this advantage. i) artificial substrates collect less debris than other sampling methods, making samplers easier to clean and sort ii) they are small, light, inexpensive and simple to construct iii) artificial substrates are easy to handle and iv) surface area is relatively easy to calculate.

Several authors proved that artificial substrates are relatively easy to clean and that samples can be sorted quickly compared to conventional methods (Hilsenholf, 1964; Barber *et al.*, 1979). However in contrast, Fredeen *et al.* (1978) and Roby *et al.* (1978) reported, that debris may accumulate on the samplers in some locations, leading to increased sorting time. Sorting time may also vary with type or size of artificial substrate and may be greater for artificial substrates than for direct sampling methods (Rabeni and Gibbs, 1978).

VI. Permit non destructive sampling of an environment.

Unlike the Ekman grab or Surber sampler, artificial substrates do not remove any thing from the substrate in large quantity.

2.4.2 Disadvantages

I. Colonization dynamics incompletely known

This is the most serious disadvantage in the use of artificial substrate as a macroinvertebrate sampling method. Factors which determine the colonization and distribution of aquatic invertebrates generally appear to be complex and difficult to study (Shaw and Minshall, 1980).

A number of factors are involved in the colonization of artificial substrates. The initial step is that the colonist first contact the new habitat. So placement of the artificial substrate, e.g. floating, suspended in water column or buried in substrate, is extremely important. Disney (1972) pointed out several factors which affect colonization. These are i) density of the source population ii) relative abundance of other suitable substrates iii) exposure time of the artificial substrates (iv) intensity and nature of the factors causing an individuals to move from its previous location v) acceptability of the artificial substrates.

The fauna colonizing an artificial substrate usually can not be considered as representative of the natural habitat because of the selectivity of the artificial substrate. However, selectivity may be a definitive advantage in studying the life histories of certain taxa (Crossmen and Cairns, 1974; Macan, 1977). As compared to other sampling methods, seasonal variation in sampling by artificial substrate also occurs.

Roby *et al.* (1978) used ASS made up with 5 cm diameter porcelain balls filled in wire mesh basket in small Northern Californian streams to study the factors that affect colonization by macroinvertebrates. They found most variation in numbers of organisms correlated with increases in organic detritus with time as a second most important variable.

Drift has been found to be an important factor in recolonization of natural substrates and is probably an important factor in colonization of ASS as well. Recolonization occurs mostly by drift and upstream movement of invertebrates (Williams *et al.*, 1976). As cited in Cover *et al.* (1978) Elliott and Minshall (1968) reported that nearly all species taken in bottom samples were represented in the drift.

II. Artificial substrates require long exposure time to obtain a sample.

This is a typical feature for passive sampling methods and the design of the sampling program should take this requirement into account.

Sampling macroinvertebrates using 5 cm diameter porcelain balls covered with wire mesh, optimum exposure period of 2-4 week was recorded (Roby *et al.*, 1978). Using floating plastic webbing as a substrate, Dickson and Cairns (1972) found a maximum number of species and total organisms in the 5th week. Mason *et al.* (1973) tried to assess the performance of ASS, using depth and exposure period as variables. They concluded that time was significant in controlling the number of organisms collected per sampler.

Arthur *et al.* (1969), used ASS made with pressed Masonite board to evaluate water quality in the Minnesota and Mississippi rivers. The benthos recovered from the Mississipi river ASS reaffirmed the zones of water quality defined by results from conventional bottom sampling techniques, while animals collected from Minnesota river ASS differed from conventional bottom sampling techniques. They suggested that the factor limiting river biota can be a lack of suitable natural habitats rather than chemical quality.

III. Loss of fauna on retrieval of samplers

Although it is expected that species lacking adaptation for "clinging" such as bivalves, crustaceans and highly active groups such as Ephemeroptera would be lost when artificial substrate are retrieved, Mason *et al.* (1973) reported conflicting results. However, some authors recorded that significant losses do occur when the artificial substrates are retrieved (Rabeni *et al.*, 1978; Pearson and Jones, 1975). Rosenberg and Wiens (1976) found that 19% of Plecoptera 61% of Ephemeroptera and 16% of Tricoptera were washed into the dip net during retrieval of rock-filled baskets from two Canadian rivers.

IV. Unforseen losses of artificial substrates

Artificial substrates are vulnerable to extreme water level fluctuations, either high or low water levels or drought, burial through sedimentation and vandalism (Disney, 1972; Mason *et al.*, 1973) which can occur during the long exposure period required to obtain sample. Roby *et al.* (1978) mentioned some practical difficulties they met while sampling macroinvertebrates using ASS, such as samplers lost, clogged, buried and predation by fish on the macroinvertebrates.

Rosenberg *et al.* (1982) suggest the following general strategies to overcome this disadvantage. i) collect geographical and climatic data from the watershed to assest suitable spatial and temporal placement of samplers ii) use an adequate number of replicates made of inexpensive materials to compensate for lost artificial substrates.

V. Inconvenient to use and logistically awkward.

Passive samplers generally require two visits, one to set the samplers and one to recover the samples (Hellawell, 1978). If artificial substrates are used in a routine surveillance program, this duplication can be minimized, because collection of the exposed samplers and setting out new ones can be done simultaneously. Problems regarding the handling of artificial substrates have been cited by several authors. Thus, samplers can be cumbersome, heavy and awkward to store and transport (Beak *et al.*, 1973), installation can be difficult and may require appreciable time (Hellawell, 1978), and samplers may require appreciable time and effort to find, recover and clean (Beak *et al.*, 1973).

Because of the frequent use of different types of ASS, several studies have been made to determine which type is more efficient in collecting macroinvertebrates. In addition, some authors have compared ASS with conventional methods. Fulner (1971), compared the taxonomic composition of fauna collected with multi-plate sampler and barbecue basket. He found them to be similar. Mason *et al.* (1973) found that if enough surface area is exposed with multi-plate samplers the results are comparable to basket samplers. Beak *et al.* (1973) reported that frequency of multi plate sampler and wire mesh cage sampler were similar.

Welton *et al.* (1982) compared two types of colonization samplers with a conventional method (core sampler) for quantitative sampling of macroinvertebrtes in the gravel substratum of an experimental recirculating stream. They found no significant difference between the number of taxa collected by each method, and significant differences in the densities of individual taxa were found in only 3 out of 64 comparisons. Roby *et al.* (1978) compared basket type ASS with Surber samplers and found a significant difference in the average number of animals collected. They suggested that could have been due to differences in the surface areas of the two sampling methods.

GEMS, the Global Environmental Monitoring System, is a United Nations organization concerned with gathering the data necessary for the effective management of the environment. Within this frame work GEMS/Water was set up by collaboration between UNEP/WHO/UNESCO/WMO, and is concerned with collecting the data necessary to determine the efficacy of various Rapid Bioassessment Protocols (Thorne, 1993). The standard nature of artificial substrate samplers certainly makes them appealing for use in the field as standard samples using dredges or grabs are hard to achieve.

Thorne (1993) carried out research to test the RBP protocol in the River Treant. He used standard artificial substrate samplers, made of biological filter medium and dredge samplers as a conventional method. Most of the biotic indices calculate for the artificial substrate samplers ranked the sites in order as expected. Further, the artificial substrate samplers captured more families and individuals than the dredge, and the differences between sites were more pronounced.

2.5 Data analysis

Some studies will need only the simplest presentation of data by graphical or tabular means while others will require statistical analysis. The interpretation of biological survey data is essentially a series of comparisons of spatial or temporal data or both and a variety of methods of data analysis are available to facilitate the process (Abel, 1989). The raw data collected from any research study can be condensed by using it to compute one or more of several numerical indices or coefficients. Four main types of such indices or coefficients are used- pollution indices, biotic indices, diversity indices and similarity indices or coefficients (Abel, 1989).

Pollution indices have reached a high degree of sophistication in many central and eastern European countries (Sladecek, 1979 as cited in Abel, 1989). They are essentially based on the descriptive saprobean system of Kolkwitz and Marsson (1909), and concern the fact that in rivers subject to organic pollution, communities downstream of the pollutant input show a regular and more or less predictable sequence of changes in the presence and abundance of indicator species.

Diversity indices were developed by theoretical ecologists and can give an idea about the spatial and temporal variation in diversity in ecosystems (Abel, 1989). Ecosystem diversity is not easily defined and therefore can be measured in several different ways. Washington (1984), as cited in Abel (1989), listed no fewer than eighteen different diversity indices used in water pollution monitoring studies. A practical advantage of diversity indices is that it is not necessary to name specimens and thus it is a useful method of analysis in circumstances where taxonomic skills are lacking or where it is necessary to work with unfamiliar groups of organisms.

Several biotic indices exist for water quality monitoring works such as the Trent biotic index, Chandler biotic score, Biological Monitoring Working Party (BMWP) score etc.. Generally the Chandler biotic index has been found to be among the most satisfactory and may be recommend as a good method for routine assessment of water quality (Abel, 1989). The BMWP score system in Britain has the advantage of taxonomic simplicity and is applicable to a wider range of geographical areas than other indices (Armitage *et al.*, 1983). Sannarm (1993) used it unmodified in Thailand with a degree of success and it has been used in India where it had gradually been modified to suit the local fauna. In the BMWP score system, each family has a score from 1-10 reflecting its general tolerance to organic pollution and the scores for the all the families present are totaled at each site. The higher the values the better the water quality. Another index the Average Score Per Taxa (ASPT) is calculated by dividing the BMWP score by the number of scoring families. It is less sensitive to sampling variation and seasonal factors than the BMWP score.

Analyses of collected data by similarity indices offer some important advantages and may eventually be found superior to analyses by diversity indices (Abel, 1989). Comparisons may be made simultaneously in space and time, each site or sample being compared in turn with every other site or sample. The simplest level of comparison is the species composition of the community. Several indices are available, but only those which include joint presence e.g. Jaccard coefficient, Sorensen's coefficient are important (Hellawell, 1986).

Another useful method for comparing communities is the use of distance measures in which the relative abundance of species within compared communities are represented in an multi dimensional space. The spatial separation or "distance" between these communities provides a measure of their affinity. In addition, communities may be compared by means of ranking methods in which species are ranked according to the relative importance in each community and then compared (Hellawell, 1986). Suwanrat (1996) incorporated several biotic, diversity and similarity indices in a rapid bioassessment protocol used to investigate water quality.

The Institute of Fresh water Ecology (IFE), in U.K., developed a software package, River InVertebrate Prediction And Classification System (RIVPACS), which can be used to generate site - specific prediction of the macroinvertebrate fauna to be expected in the absence of major environmental stress (Wright, 1995).

Data transformation is need before performing parametric statistical analysis if raw data are not normally distributed or variance are not equal. If replicates collection have been taken at each site or time, a plot of sample variance against the sample means, followed by a test of correlation between two will indicate whether transformation is necessary (Rosenberg *et al.*, 1993). Among different transformation methods, square root transformation is applied if the variance and mean are approximately equal and a log transformation is applied when the variance is consistently greater than the mean.