Proposed Biotic and Habitat Indices for use in Kansas Streams

Report No. 35 of the
Kansas Biological Survey
The Universtiy of Kansas, Lawrence, KS 66045

February 1988

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Support for this project was provided cooperatively by the Kansas Department of Health and Environment and the Kansas Biological Survey under KU Acct. No. 5464-x705.

Second printing (electronic reformatting), November 2003

ACKNOWLEDGEMENTS

We would like to thank the Biological Survey staff for their helpful suggestions, comments, and contributions to this project. Assistance from Paul Liechti, Len Ferrington, Alex Slater, Franz Schmidt and Cory Koeppen were invaluable and greatly appreciated. We give special recognition to Judy McPherson for her highly skilled technical assistance in completing the final manuscript.

The direction provided us by the Water Quality Assessment staff of the Bureau of Water Protection (Kansas Department of Health and Environment) added much to the success of this study. We are indebted to Donald Snethen and Joe Arruda, both of KDHE, for their patience and guidance through these efforts.

We especially wish to thank Dr. Ed Martinko, director of the Biological Survey, for providing the additional funding required to complete this project in the comprehensive manner that we, as scientists, felt was necessary to me*et* all study objectives.

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AN INTRODUCTION TO BIOTIC INDICES

In the study of water pollution and the related “health” of aquatic ecosystems, three general approaches have found universal appeal: indices of diversity, similarity indices and biotic indices. The primary purpose of this section is to review, discuss, and evaluate proposed biotic indices. General comparisons among the three general evaluation approaches are made when appropriate. All discussion refers to the use of macroinvertebrates in lotic aquatic environments. A more thorough discussion of the comparative merits of diversity, biotic and similarity indices can be found in Washington (1984). Several new indices have been proposed since the publication of Washington’s review and some existing indices have been modified. All are attempts to improve the basic usefulness of a biotic index in identifying biological change often associated with anthropogenic environmental impacts on aquatic systems.

There are basic differences between biotic indices and diversity and similarity indices although all are often used to indicate stress or changes in biological communities. Indices of diversity and similarity are quantitative measurements of total community structure. Diversity indices can be used to assess biological quality of various aquatic environments by giving a measure of the structure of the total macroinvertebrate community at each site. A similarity index also uses total community structure parameters, but unlike a diversity index it cannot give a value for a single site. Similarity indices are comparative measurements and can only indicate similarity of the structure of two communities. Evaluation of many sites is only done by making all possible paired comparisons, thus comparisons among different sets of similarity indices cannot be made. Unlike a similarity index, a biotic index can be calculated for a single community and can be compared to diversity indices, other site specific parameters, and values from other studies. However, a biotic index does not measure total or “true” community structure. Biotic indices are based on the “indicator organism” concept. A biotic index value for a community is a measure of the physiology, toxicology and ecology of the organisms that “indicate” absence or presence and often the degree of particular impacts. A biotic index is weighted by the mortality or survival of various “indicator” organisms from specific taxa and trophic levels within the community. Thus, diversity, similarity and biotic indices use different approaches to give numerical descriptors to biological communities. Furthermore, they have all been applied to evaluate water pollution impact, yet only the biotic index was designed to discern particular type(s) of ecological impacts.

There has been both support for and criticism of the use of biotic indices and diversity indices for assessments of pollutant effects. Wilhm (1970) and many others have argued that diversity indices are useful measures of the responses of aquatic communities to pollution. Cook (1976) investigated the usefulness of the Shannon-Wiener diversity index as a measure of pollution. Based on her own work and that of Mackay *et al*. (1973) and Harrel and Dorris (1968), she concluded that this diversity index may be useful only for indicating relatively large inputs of pollutants and thus was not reliable for a continuous assessment of increasing or decreasing water quality. In Cook’s 1976 study involving direct comparisons of various pollutant measures (diversity and biotic indices), she stated that “the average Chandler score (a biotic index) is most sensitive to variables influenced by pollution” (organic), and “it is least likely to be influenced by seasonal changes or sample size and thus most likely to give a continuous assessment of water quality.” Critics of biotic indices are quick to point out that indicator species are often sensitive to one pollutant and tolerant to another. Cairns (1977) notes that the indicator organism approach has many weaknesses, one of which is undoubtedly this. Myslinski and Ginsburg (1977) felt that selection and categorization of indicator organisms is subjective and depends on the knowledge and experience of the biologist. This makes different biotic indices difficult to compare. At least some of their concerns often apply to other assessment approaches and the need for comparability between biotic indices may be of minimal importance within regional applications. Lawrence and Harris (1979) also voiced concerns about the often subjective manner in which tolerance values are assigned and offered a quantitative method for ranking water quality tolerances of benthic species, but even their approach contained somewhat subjective research elements. It should be noted that many of the proposed biotic indices list tolerance values for indicator organisms that are in one sense subjective values but the selection of these values were based on sound and often very comprehensive empirical data (*e.g.*, Chandler 1970; Chutter 1972; Hilsenhoff 1977, 1982, 1987).

One recurring limitation of biotic indices discussed by supporters and critics of biotic indices is that they should not be considered to have worldwide applicability. Many species are not ubiquitous, thus taxonomic composition will vary widely as will indicator organisms. Investigators' interpretations of sensitivity are often based on local conditions. It seems that no single biotic index and associated tolerance value list will work in every state or country in the world. Given this, biotic indices are likely to be geographically specific. In 1972 the U.S. EPA reported that the use of indicator organisms (such as in biotic indices) was not commonly accepted. However, over a decade earlier, King and Ball (1964) stated that one of the most generally accepted biological assessment techniques is that of using indicator organisms. The latter statement is an easily defended one, if one examines the literature carefully and takes into account the nearly universal use of indicator species approaches outside the United States. The use of indicator organisms has certainly grown steadily both within and outside the U.S., especially among water control, regulatory, and research authorities.

The "indicator organism" concept forms the basis of all biotic indices. Indicator organisms are test species picked for their known sensitivity or tolerance to various parameters, usually organic pollution, or other types of pollution (*e.g.*, heavy metal pollution).

Chandler (1970) commented that the concept of an indicator organism whose presence proves pollution is incorrect. Often these "indicator" species may also be found in clean streams. He maintains that in clean streams there is usually a diverse fauna where the percentages of the total numbers in each species will be low and similar, but in polluted situations the fauna will be restricted and tolerant dominants will appear. There is general agreement that organic enrichment tends to restrict the number of species and simultaneously increase the numerical abundance of tolerant species (*e.g.*, Bartsch 1948; Hynes 1960; Mackenthun 1969). While tolerant organisms may become dominant in polluted environments they can also be found in a variety of water quality conditions. However, sensitive organisms by definition are restricted to clean or cleaner water. Thus, it is within the sensitive group of indicator species that the most valuable assessment information is to be found.

Lewis (1978) contended that the expectation of changes in number, biomass or growth of a species will reflect the species response to pollution only if among several other conditions the species was autotrophic and living virtually alone with only the physical and chemical environment to respond to. Furthermore, he says, it is naive to identify and count every individual present in an ecosystem unless one has an understanding of the interactions between all species capable of existing in that environment. He assumes that “key” species are more sensitive to pollution in general than other species. Lewis claims that if the “key” species succumb then the community will inevitably be altered, if they survive so will many others. In general his views are very supportive of the indicator organism approach.

Scientifically there appears to be no single best approach of measuring the biological change (impact?) that may be brought about by man-induced water pollution (Washington 1984). Often the “best” approach to the biological evaluation of water pollution becomes dependent more on local regulatory needs, regional environmental quality, available resources and expertise. In a recent evaluation of potentially useful biotic and water quality indices for use in Kansas, biotic indices were highly promoted because of the state’s need to monitor very different types of streams (*e.g.*, sand-bottom rivers, pool-riffle streams); to assess the impacts of both point and non-point pollution; and to perform biological assessments throughout the state despite limited state resources (WAPORA 1984). Only a few biotic indices were reviewed in this study but recommendations were made to investigate the potential of modifying an existing biotic index (*e.g.*, Hilsenhoff’s biotic index) to be used specifically for Kansas. The use of a specific biotic index for this region is in keeping with U.S. EPA's current emphasis on regionally based water quality programs and criteria development.

A REVIEW OF OTHER BIOTIC INDICES

The following is a review of existing biotic indices that utilize macroinvertebrates as indicators. The advantages and disadvantages of different taxa as indicators of aquatic pollution have been well summarized by Hellawell (1977a). Based upon the biological assessment needs of Kansas, we concur with Hellawell's findings that macroinvertebrates, in general, are "better" indicators of the biological health of flowing water in regards to water quality conditions than other biotic groups. The adoption of macroinvertebrates (and a biotic index based on their use) is, therefore, recommended for the biological monitoring of water quality in Kansas, on the following grounds:

1) high public visibility;
2) past history of successful use in Kansas;
3) availability of identification keys for most taxonomic groups;
4) a high "hysteresis value", because of their sedentary or relatively stationary habits and long life cycles, which allow meaningful spatial analyses of results and make temporal analyses possible; and
5) heterogeneity, *i.e.*, several phyla are represented which increases the probability that at least some groups respond to a given environmental change.

We reviewed the following biotic indices in an attempt to evaluate their potential for use in Kansas streams. Evaluation was directed to those properties outlined by Cook (1976) as generally being desirable qualities of a pollution index. They are:

1) use as a continuous (linear) assessment from unpolluted to polluted conditions;
2) sensitivity to the stressful effects of pollution on the aquatic community;
3) independence from sample size;
4) general application to various types of streams; and
5) ease of data collection and calculation.

In many cases not all aspects of a particular index can be evaluated because of insufficient published or unpublished information on some indices. Not all biotic indices that have been proposed and used in aquatic ecosystem evaluation are reviewed here. Many such indices are only slight variations of those covered in this report. Some of the review comments offered by WAPORA, Inc. (WAPORA 1984) in their evaluation of water quality and biotic indices for use in Kansas are integrated into this evaluation. Lastly only biotic indices which are solely based on biological data were considered in this evaluation. It was our thought that a biological index should reflect the overall impact of water quality and habitat quality and not be linked to specific physical or chemical water quality measures. Chemical and physical characteristics of normally healthy streams vary widely and this generally precludes their reduction to a simple standard or set of parameters. The worth of a biotic index which includes chemical qualifiers and the need for an index to be related to specified chemical parameters is somewhat questionable (Cook, 1976).

The Saprobic Systems

The earliest attempt to provide a index of the changes observed in aquatic communities in response to pollution (organic enrichment) was the "saprobien system" which has been modified, developed and expanded over the last 50 years by many workers. It is beyond the scope of this work to provide a comprehensive treatment of the various saprobien based indices. Excellent reviews of these systems may be found in Sladecek (1973) and Persoone and DePauw (1979).

Saprobity is the state of the water quality resulting from organic enrichment as reflected by the species composition of the community. It was developed through the pioneer work of Kolkwitz and Marsson (1902) who eventually detailed a "saprobic system" of zones of organic enrichment and a classification of a wide variety of species (traditionally including algae, ciliates, flagellates, rotifers, microcrustacea, insects and even fish) that lived in different saprobic zones. This is the first measurement that can be considered a biotic index. The different zones of degradation were: polysaprobic, alpha-mesosaprobic, beta-mesosaprobic and the oligosaprobic zone. Chandler (1970) claims that the saprobien system can not be used to evaluate short turbulent streams or rivers receiving poisonous or non-biodegradable waste. Both Chutter (1972) and Hynes (1960) were critical of its limited usefulness (*i.e.*, organic enrichment of large rivers) while Hynes further noted that it was unwieldy to use, failed to account for local influences and depended on the identification of microorganisms. Certainly its lack of adaptability to stream size and type, limits its potential value in regional or other comprehensive water quality assessment programs.

Oligochaete Indices

Several indices were developed over the years that utilized aquatic earthworms (Oligochaeta) as indicator species. Wright and Todd (1933) used the total density of oligochaetes to assess the degree of pollution based on various worm densities. Later Goodnight and Whitley (1960) suggested that the relative abundance of oligochaetes to all other benthic organisms be used as an index of pollution. Actually only tubificid or "sludge worms" were considered and the index appears as:

$$\frac{\text{number of tubificids}}{\text{number of all other organisms}} \times 100$$

This index becomes dependent on the presence and dominance of *Tubifex* and necessitates the enumeration of all organisms collected.

Unknowingly, King and Ball (1964) developed a simpler version of the Goodnight and Whitley index by replacing organism abundance with weight. Their index is the ratio of aquatic insect weight to tubificid weight. The $\log_{10}$ of this ratio was then plotted against distance from point source impact, thus the index equals:

$$Log_{10}\left(\frac{\text{insect weight}}{\text{tubificid weight}}\right)$$

The main advantage of this index appears to be that little taxonomic skills are required to use it. The authors state that it did identify both domestic and industrial (heavy metals from a plating facility) pollution.

Another example of the use of aquatic oligochaetes as indicators was the index proposed by Brinkhurst (1966). In this index Brinkhurst used the number and proportion of *Tubifex* and *Limnodrilus* species to all other species to indicate organic enrichment. Hellawell (1977b) was critical of nearly all these oligochaete indices referring to some as both crude and naive. For whatever reasons none of the oligochaete indices were ever used to any extent and their worth remains in question. It is our opinion that certain oligochaete species are good indicators and their use in broader based biotic indices would be beneficial, however, many species are tiny and fragile which makes collection and preservation a problem. In addition identification of oligochaetes to the species level is often limited to sexually mature specimens and nearly all specimens should be slide mounted and cleared to facilitate identification. These tasks can be costly in terms of manpower and time.

Beck's Biotic Index

Working in Florida, Beck (1955) devised a rather simple index using freshwater macroinvertebrates to estimate the impact of organic pollution. He initially considered only a single value representing a faunal evaluation of impact based on the combination of clean water species (Class I organisms) and the total number of species within the stream in question. However, he abandoned that concept because he felt that high species number reflects diverse habitat rather than clean water. Beck claimed that the above procedure did not take into account organisms tolerant of moderate levels of organic pollution (Class II organisms) which do reveal something with regard to water quality. He offered no more explanation concerning his definition of Class II organisms. He soon recognized that if species numbers of Class I and II were added there was a major area of overlap in the instance of low index values for certain types of clean streams with relatively low numbers of species and sometimes high indices for moderately polluted streams. Beck proposed the following formula to minimize this overlap:

$$BI = 2 \times (\text{number of Class I species}) + (\text{number of Class II species})$$

In practice he found the index values to range from 0 to 40 with clean stream values being ≥ 10; moderately polluted streams ranging from 1 to 6 and grossly polluted streams having a zero value. He noted that clean streams with limited habitat and low velocity often ranged in value from 4 to 9 and that the index was closely linked to stream velocity. While today Beck's index is still used in Florida, there are few cited studies that have utilized his work. Doudoroff and Warren (1956) and others have noted that Beck's index can only be used for organic pollution which was the condition Beck chose to identify. Of more concern is the index's apparent dependence on stream velocity making the evaluation of water quality in slow-flowing streams difficult.

Heister (1972) later modified Beck's BI by assigning all of the invertebrate community into five classes but still used only Beck's formula for the first two classes. Heister supplied a complete list of organisms. He also compared his index to a diversity index (H') and found a positive correlation between the indices.

Beak's "River" Index

Over a period of six years Beak (1965) studied the macroinvertebrate community of a large Canadian river impacted by organic and toxic pollutants. He proposed a biotic index of water quality based on the feeding habits, sensitivity to pollution and invertebrate densities (Table 1). All macroinvertebrates that are collected are enumerated and used in Beak's analysis. The index can be derived from samples obtained by any method which permits a reasonable measure of population densities. He says it is essential to include control samples from unpolluted areas for each habitat type sampled in polluted waters. Beak's index is based on the acquired knowledge of the ecology and toxicology of the organisms under study. This index requires extensive collections, high taxonomic resolution, and a comprehensive, toxicological and trophic classification database. This probably explains why the index has never been used by other workers.

While there exists some major weaknesses in the Beak river index it represents the first major attempt to incorporate a number of physiological and ecological factors into a biotic index. Chutter (1972) was most critical of Beak's index. He cited as major weaknesses the general lack of trophic information for benthic organisms, subjectiveness of assigning pollution sensitivity values to animals and the vagueness of density terms. In addition, Beak's index cannot take into account the potential for different sensitivities between organisms and various toxicants. These weaknesses may make this index difficult to apply to other study areas but his index concept is laudable.

The Trent Biotic Index

The Trent Biotic Index was first published by Woodiwiss (1964) who was employed by the Trent River Authority (England). Woodiwiss used only riffle inhabiting invertebrates of Midland rivers (England) in his index classification. Hand samples and kick samples taken with a hand net (780 micron mesh) are taken in such a way as to include material from all microhabitats. He devised a scheme in which the number of groups of defined benthic taxa was related to the presence of six key organisms found in the fauna. These organisms were plecopteran larvae, ephemeropteran larvae, trichopteran larvae, *Gammarus*, *Asellus* and tubificids plus red chironomid larvae. In practice, organisms are sorted into groups and streams are classified (10 for clean water to 0 for grossly polluted) according to the presence or absence of key groups and the diversity of fauna. This index like the saprobic system does not take into account the relative abundance of the organisms present.

Balloch *et al*. (1976) reviewed the Trent Index and listed a number of advantages and disadvantages associated with its use. Most notable advantages mentioned were ease of use and its ability to correctly classify moderate to grossly polluted waters. In general Balloch *et al*. were very critical of this index and indicated that it was not suitable for use as a criterion of water quality because of its general insensitivity to varying levels of impact, especially mildly and moderately polluted waters. When compared to the Chandler scores (CBS and ACBS noted below) the Trent index proved of little value in determining intermediate levels of pollution in rivers known to have a well defined spatial pattern from clean to grossly polluted conditions (Murphy 1978). Both Murphy (1978) and Balloch *et al*. (1976) also suggested that the Trent Biotic Index was affected by habitat quality making interpretation of the index difficult. Overall, the Trent Index appears to lack the sensitivity desired by most workers interested in assessing the degree of biological impairment associated with various levels of water quality.

BMWP "score"

In 1979 the Biological Monitoring Working Party (BMWP) of the International Standardization Organization of European countries devised a new biotic index scoring system (ISO-BMWP 1979). This working party attempted to formulate and score a system by which families of macroinvertebrates could be used as indicators of water quality (basically organic enrichment). Utilizing information from their individual experiences and work, the BMWP selected a number of defined benthic taxa and assigned tolerance or indicator values ranging from 10 (very clean) to 1 (grossly polluted) (Table 2). The BMWP score is similar to the Trent biotic index as it is based upon the presence/absence of certain fauna groups (families).

This system has been applied to various streams and stream conditions throughout Europe but evaluations of its performance are few (Armitage *et al*. 1983; Brooker 1984; Tolkamp 1985). Armitage and co-workers evaluated its performance over a wide range of unpolluted lotic sites and found its assessment value to differ somewhat between stream types. Brooker (1984) found that for Welsh rivers of similar size (upland streams), the Chandler score (CBS) and the BMWP score were highly correlated when only family data were used and that the more resource intensive Chandler score provided no better assessment that the BMWP method. However, our interpretation of Tolkamp's (1985) data suggests that the Chandler score may define a broader range of water quality conditions and that the median range of values were associated with fair to good water quality conditions which Tolkamp indicated best represented actual conditions.

Chandler's Biotic Score (CBS)

Chandler's (1970) research on the River North Esk and other Lathian rivers in Britain led him to propose a biotic index for use with other data (*i.e.*, chemical data) in assessing the condition of rivers. It is interesting to note that most current investigators concur with Chandler's premise that biological information should not replace other types of water quality data but should be used with other information to formulate an overall assessment of conditions. He felt that macroinvertebrates provide the easiest, most reliable biological estimates of water quality impact. In contrast, fish are too mobile to be water quality indicators. Protozoa react rapidly, but may also recover quickly, thus, not identifying long term impacts, and are often difficult to identify. Chandler states that riffles are the habitat to sample as that is where sensitive organisms live and the interpretation of his index in riffleless streams was difficult.

Chandler thought that a major problem with many earlier biotic indices was their failure to consider the abundance of the faunal elements. The mere presence of a single specimen of an "indicator species" could greatly alter a station's index causing many inconsistencies in the system. The phenomena of drift could account for the presence of some "indicator individuals" and certainly presence/absence data must always be viewed as having limited interpretive value. However, Chandler also realized the technical problems associated with measuring abundance accurately. In addition he concluded that absolute abundance had little use in routine river surveys and that relative abundance in terms of abundance categories would be sufficiently accurate when repeated sampling was utilized.

Given the common resource and method constraints of most macroinvertebrate surveys, it is unlikely that true values of the absolute abundance of community elements are ever derived. Generally, an extremely large number of samples are required to provide reasonable population estimates (*e.g.*, Hales 1962; Edwards *et al*. 1975; Hynes 1970; Resh 1979). Such efforts are usually beyond the resources available to even quantitative surveys.

Chandler formulated his index around the faunal groups of Woodiwiss (*i.e.*, the Trent index) and the "levels of abundance" used by the Lothians Purification Board (Chandler 1970). The levels of abundance used by Chandler were: Present (1-2), Few (3-10), Common (11-50), Abundant (51-100), and Very abundant (>100). He arranged organism groups in order of their tolerance to organic pollution and assigned a score (weighing factor) based on abundance to each entry (Table 3). A station score is calculated by identifying and enumerating all taxa present and scoring each group according to its abundance category. All scores are added and the station score becomes this total value.

Chandler notes that there is no upper limit for the index (score) and that differences of diversity (species richness) and abundance in the clean section of the river are easily seen. He claims the index can identify a continuous gradation from polluted to clean conditions. In their evaluation of biotic indices, Balloch *et al*. (1976) reported that the Chandler score was the best indicator of water quality impacts on biological conditions. They gave index values of 0 for no macroinvertebrates present, 45-300 for moderate pollution and 300 to 3000 for mildly polluted to unimpacted conditions. Balloch *et al*. (1976) also noted that: 1) the score had sensitivity comparable to a diversity index; 2) worked well for slow moving rivers as well as alternating pool/riffle streams; 3) the index could classify a broad range of conditions, and 4) the score was somewhat lower in a headwater stream. However, they were concerned about some of the assigned tolerance values and felt that the data was difficult for non-biologists to interpret.

Murphy (1978) claims that the Chandler biotic score is highly dependent on the number of species taken in the sample. He also noted that the score dips in headwater streams even though they were unpolluted. Hellawell (1978) considers the CBS to be the most satisfactory biotic index he assessed. Several have recommended the modification of the CBS to the average Chandler biotic score (ACBS).

Average Chandler Biotic Score (ACBS)

Balloch *et al*. (1976) and Cook (1976) both proposed that by dividing the CBS score (for a given station) by the number of taxa (Chandler's groups) present in the sample, a score would be obtained that was more reflective of water quality and less affected by natural stress. This modified or average Chandler score (ACBS) can be expressed by the formula:

$$ACBS = \frac{\sum_{i=1}^{G} \text{weighted scores}}{G}$$

where, G = number of Chandler's groups. They felt that natural stresses associated with headwater streams (*e.g.*, temperature, water velocity, substrate) accounted for the dip in the CBS and that their modification would adjust for these conditions and give values commensurate with "water quality". It is important to remember that dividing by the number of faunal groups present will lessen but not remove the group number effect, because in the CBS each group score is already weighted for group effect. The number of groups (G) does not change the fact that the weighted scores were due to particular groups. In practice what this often accomplished was to adjust the CBS score downward if the total sample score (CBS) was high due to the presence of a large number of low scoring groups (tolerant groups) that collectively inflated the overall score.

Both the CBS and ACBS were developed to identify only the effects of organic pollution, however the ACBS scores were less affected by natural occurring stresses (Balloch *et al*. 1976). Of all the diversity and biotic indices examined by Murphy (1978), only the CBS displayed both a reduction in temporal variability and a consistent spatial discrimination of sites from unpolluted to highly polluted. Most diversity indices (*e.g.*, Shannon-Wiener index) showed such marked temporal variations as to completely mask any spatial pattern while both the Chandler and Trent indices were affected in headwaters. As Washington (1984) suggests, another shortcoming of these (and other biotic indices) is that other lists of grouped taxa and their tolerances would have to be developed to assess other pollutants.

Chutter's Index

Chutter (1972) developed a biotic index for use in South African rivers based on responses of macroinvertebrate species (or taxon groupings) to organic pollution. His empirical index was established on three hypotheses concerning the stream fauna. 1) The faunal communities of unimpacted lotic waters are definable; 2) they change in a predictable way as organic material is added; and 3) the greater the amount of oxidizable organic matter added, the greater the faunal change.

Chutter qualified the predictability of his index by stating that the index only applied to riffle communities and that the index was not reliable after flood events. This index utilizes several phyla of macroinvertebrates but excludes cladocerans and copepods which Chutter said tended to drift into an area from upstream sites. Chutter drew up a list of riffle taxa and then used the literature to assign each taxon a quality (tolerance) value. Clean water species were valued at 0 and polluted species at 10.

Originally the index was derived by recording each individual organism with its quality value on the 0-10 scale, summing these up, then dividing by the total number of organisms in the sample. However, Chutter soon recognized that several taxa often present in extreme numbers tended to overly influence the final mean quality value. He added a sliding scale by which certain animals (especially Oligochaetes, and certain chironomid, simuliid and ephemeropteran larvae) were assigned a specific quality value established by its relative abundance or percent composition of the total faunal number. The final biotic index formula used by Chutter was:

$$\text{Chutter's index} = \frac{\sum_{i=1}^{n}\left(n_i \times Q_i\right)}{N}$$

where, Qi = quality value from his table and/or sliding scale for taxa i

k = number of taxa with quality value not 0

n = number of individuals of taxa i

N = total number of individuals in the sample

Chutter felt that his index should correlate with various chemical qualities of water. He implied that chemical quality equates directly with water quality, although he offered no definition of “water quality”. He was among the first authors to attempt an interpretation of river cleanliness based on the biotic index value (Table 4).

Chutter’s Index represents a somewhat newer approach to a biotic index, despite its similarity to past indices. Washington (1984) states that it is strongly an “indicator species” type of index and does not contain a true community structure approach (*i.e.*, total species diversity).

He erroneously concludes that Chutter's index does not take into account abundance as did Chandler's score, except in Chutter's use of sliding scales. Chutter's sliding scales are used to adjust the quality values of certain taxa by taking into account the relative abundance and/or number of species of other taxa found in the same sample. In fact the number of individuals of each taxa and the total number of individuals comprising a sample are used to calculate the sample value for Chutter's index.

Pinkham and Pearson (1976) criticized Chutter's index because it offered no measure of similarity and thus could have identical values for totally different communities. Chutter (1978) responded by pointing out that it was not developed to measure similarity and it was acceptable to derive the same values for different communities if both were responding to similar degrees of organic richness. He also noted that it has proven to be very useful in South Africa (*e.g.*, Coetzer 1978).

Hilsenhoff's Index

Hilsenhoff (1977, 1982, 1987) was apparently the only worker outside of South Africa to either use or examine the potential use of the Chutter Index for aquatic systems elsewhere. The initial index proposed by Hilsenhoff differed from Chutter's Index in the following respects:

1. Organisms were assigned a "quality" value ranging from 0 to 5 (not 0 to 10).
2. Only aquatic insects, isopods and amphipods were given quality values. No Culicidae, Dixidae or Stratiomyidae larvae; no Hemiptera; no Coleoptera other than Dryopoidea; and no arthropods < 3 mm long except adult Elmidae and mature Hydroptilidae larvae were used in his final index scheme.
3. Taxonomic level identifications and "quality" value assignments were supposed to be at the species level.
4. Samples were to be obtained by using a timed collecting effort. However the biotic index formula remained basically unchanged:

$$\text{BI} = \frac{\sum_{i=1}^{k}\left(n_i \times a_i\right)}{N}$$

where, n = numbers of individuals of taxa i

a_i = tolerance value assigned for taxa i

k = total number of taxa

N = total number individuals in the sample

Hilsenhoff claimed that his index provided an estimate of the degree of saprobity and possibly trophism of a benthic population (Hilsenhoff 1977; 1982; 1987). It should be noted that he only utilized riffle communities in his assessments. His justification for using only insects (excluding those families mentioned in 2 above), amphipods and isopods is that they are generally abundant, easily collected, they are species rich, not mobile and most have a one year or longer life cycle. Hilsenhoff's first quality values were empirically derived for various organisms after several years of study on 53 Wisconsin streams exhibiting various levels of organic enrichment (1969-1973). Hilsenhoff limited his sample size to 100 individuals, or less (if 100 arthropods cannot be found in 30 minutes of sampling and picking). He originally suggested that a maximum of 25 individuals be used for any one taxa (1977) but later dropped the idea (1982).

Utilizing this index on Wisconsin stream data Hilsenhoff proposed that a series of stream water quality conditions could be identified (Table 5).

Hilsenhoff identified and quantified the temporal effects on index values and offered a correction factor for seasonal differences. He recognized the value of species identification especially when species in a genus may differ greatly in their response to an impact. He does use generic values when all species within that genus are known to have similar responses and promotes the use of generic values (when possible) because of the reduction in identification time.

While Hilsenhoff did not find it necessary to use limiting abundance categories or a sliding scale to modify the effects of highly abundant organisms, he controls the number and abundance of organisms that constitute a sample. By not considering organisms smaller than 3 mm in length he effectively avoids using the often abundant young instars of all arthropods. Hilsenhoff (1987) recommended that a sample be collected with a D-frame aquatic net and that the collector should:

1) collect in current (>0.3 m/sec) preferable in a riffle;
2) avoid collecting from rooted macrophytes and filamentous algal mats; and
3) collect until there is enough debris in the net to fill an 8-ounce jar or it is obvious that more than 100 arthropods have been taken.

In his latest biotic index publication (Hilsenhoff 1987), the assigned tolerance values were expanded from the original 0-5 scale to 0-10 to accommodate intermediate values derived from new data. Additional data from more than 2000 samples collected from over 1000 streams during 1979-1980 were used to re-evaluate tolerance values and to expand the tolerance scale. New tolerance values were assigned 359 species and/or genera found in streams examined in his work.

Hilsenhoff states that his index is rapid, sensitive and reliable but several problems may complicate its interpretation. The need for keys to species; influence of stream current and temperature, seasonal changes, and impact of habitat variables are some of the problems that need to be addressed to make his index more functional. For instance, seasonal difference in biotic index values were often found to be statistically significant and can jeopardize interpretation of results (Hilsenhoff 1982).

We must conclude that Hilsenhoff's biotic index functions extremely well in identifying specific types of organically enriched streams in Wisconsin. The very large database used to derive his empirical tolerance values, the similarity of specific habitats sampled and the selective exclusion of various groups of arthropods probably contribute to the success of this index but at the cost of making it a very restrictive one.

Belgian Biotic Index

DePauw and Vanhooren (1983) described a biotic index based in part, on the Trent biotic index (Woodiwiss 1964) and the work of Tuffery and Vernaeux (1968) which has proven very successful in the Belgian Water Quality assessment program. The index has been widely field tested in Europe using the results of over 5000 benthic macroinvertebrate samples collected from over 30,000 km of stream and river reaches. While never stated, this index appears to have been developed to measure changes resulting from organic enrichment.

This index is calculated from data on the presence or absence of selected taxonomic groups referred to as “systematic units” (SU). Thus the level of taxonomic identification varies between taxonomic groups as defined in Table 6.

Samples are processed in the laboratory and selected groups of organisms classified into systematic units according to Table 6. The biotic index is then derived from a standard table (similar to the Trent biotic index table) developed originally by Tuffery and Verneaux (1968). The index is determined by the presence of faunistic groups (Column I), the number of systematic units of that group and the total systematic units that constitute a sample (Table 7). Seven faunistic groups are ranked according to pollution sensitivity. Increasingly tolerant taxa are placed sequentially in groups 1-7 down Column I of Table 7. The determination of the index is dependent not only on the number of systematic units present but also on what systematic units are absent. The index is derived from the table by first selecting the most sensitive faunistic group present in the sample. For example, if taxa from groups 2 and 3 are present use faunistic group 2 for the next selection. If group 1, 2, or 3 is present, the first or second row of Column II is chosen according to the number of SU of that group that are present. Then in Column III one selection is made which corresponds to the total number of SU present in the sample as noted at the top of column III. This final selection now includes all the SU present in the sample, even if they are from a more tolerant faunistic group. The crossing of the selected row and column determines the final index. The values of the index may vary from 0 to 10. The index assumes that the presence of two genera of Plecoptera or Heptageniidae (Ephemeroptera) and the presence of a total of 16 or more systematic units is an indicator of unimpacted conditions.

Benthic samples are collected at each site in a standardized manner using a D-frame net with a mesh size of 300-500 microns. The collecting technique is designed to determine as accurately as possible the species richness and types of organisms present at each sample location. The sampling is not confined to riffles but all accessible microhabitats in all habitats are sampled (*e.g.*, stones in currents, aquatic vegetation, mud on pool bottoms). Sampling efforts are somewhat limited (3 to 5 minutes) considering the necessity of sampling all available habitat types.

Lafontaine *et al*. (1979) and DeBrabander *et al*. (1981) tested the index and reported excellent results in determining appropriate water quality conditions. It is important to note that they found the index exhibited little variation in determining water quality despite the differences in species composition resulting from habitat variability between and/or within stream reaches sampled. DeBrabander and DeSchepper (1981) compared the use of biotic (including the Belgium index) and chemical indices in Belgium and concluded that chemical indices displayed high temporal variability. This natural variation in chemical quality can only be defined after a large number of chemical measurements are made over an extended time period.

WAPORA (1984) cite two problems associated with the Belgium biotic index. The first centers around the problem of estimating the degree of pollution because of the "large number of variables which may affect the value of the index". They do not elaborate on this but mention that establishment of suitable reference (unpolluted) ecosystems could be used as a basis of comparison with other areas. While we agree that one must be able to define good to determine bad, the Belgium index appears to relate well with diminished biological quality and only an interpretative classification scheme (similar to Hilsenhoff's (Table 5) or others) needs to be worked out. The second problem pointed out by WAPORA was the lack of a suitable sampling technique when a D-frame net cannot be used. Recently DePauw, Roels and Fontoura (1986) reviewed the results of three years of experience in Belgium and Portugal with artificial substrates for collecting organisms used in water quality assessment by means of the Belgium biotic index. They site artificial substrates as providing a valid alternative method for sampling the macroinvertebrate fauna and indicated the possibility of their use in standardizing the sampling effort. It was stated that sampling with a handnet may be more subjective, that is, causing more variability due to the collectors.

It should be emphasized that the Belgium biotic index is based solely on the use of presence and absence data and the implied tolerance values associated with the systematic units utilized in the scoring scheme. Thus the presence of a single individual within a sensitive faunistic group can cause the index value to increase two or more points (≥ 20% increase). This factor would appear to make this index overly sensitive to drift where drifting organisms may be brought into a collecting area from unaffected upstream areas. This index is based strictly on the indicator species concept and the numerical distribution of the community sampled is not considered. However, the significance of abundance distributions of sensitive and tolerant organisms are not easy to interpret. Thus, excluding abundance information may or may not be a disadvantage.

The Belgium index is noteworthy in that it has not been restricted to use in riffles or other specific stream habitats. It is to be used with standardized collecting techniques which maximize the species richness of the sample of any stream type. Examples include exploiting all macro- and microhabitats at a site. Provided information is available concerning each organism's (or group's) response to pollutants, the advantage of maximizing the species composition of a sample is to yield more tolerance data about the community. This increases the information base that ultimately contributes to the index value.

Summary of reviewed biotic indices

It can readily be seen that many of the cited biotic indices possess distinct characteristics, but all are based on the concept that various organisms have identifiable degrees of tolerance to specific pollutants, pollution conditions (*i.e.*, organic enrichment) and/or environmental factors.

This is in essence the “indicator species” concept. All indices attempt to distinguish between anthropogenic and natural stresses and all try to define “water quality” with respect to various types of changes in biological populations or communities. More often than not, the definition of “water quality” is left to the reader to determine. Nearly all indices are based on the kind of biological changes that have been associated with organic enrichment. Thus, this is probably the only real “water quality” assessment that is being made with current biotic indices. There is some evidence that a few have worked successfully in discerning among sites that are known to contain various toxic substances (*e.g.*, Solbe 1977; Watton and Hawkes 1984).

All indices theoretically yield a linear ranking system of progressive values which indicate decreasing (or increasing) biological “water quality” conditions. Minimum and maximum possible values often differ and whether values are an arithmetic or geometric series is unclear. Thus, most biotic indices cannot be translated into each other. They each weight structure (*i.e.*, taxa diversity), abundance information, and biological attributes (*i.e.*, taxon tolerance and sensitivity) differently. Resulting values from different indices can only be roughly compared. The relationships between indices are also probably not linear (Tolkamp 1984; Illies and Schmitz 1980). Only a few associate biotic index values or biotic scores with a classification scheme that defines perceived degrees of water quality (*e.g.* excellent, fair, grossly polluted, *etc.*). Ultimately it is important to choose an appropriate assessment system which has been developed or modified for use under local or regional conditions and can be ecologically interpreted for regulatory and other purposes. It appears desirable that the system have a well defined maximum and specific ranges which relate to various levels of pollution.

One final consideration in attempting to use biotic indices to assess “water quality” must be addressed at this point. Like so many workers before us, we are left with the fact that too often in stream assessment situations there exists no reliable and independent reference to make evaluations against. In general, we are inclined to use physical and chemical features as a reference and to measure a deterioration of the chemical water quality parameters in a parallel classification with biological parameters. Certainly this method has been used successfully by many authors (*e.g.*, Woodiwiss 1964) but relationships between biological phenomena and chemical parameters are not always clear or even linear in nature (*e.g.*, Schmitz 1975). The situation regarding an accurate assessment of water quality in chemical terms is so confusing that it is probable that most biological classification methods more accurately reflect overall "water quality". We are left with the realization that there is no proven and reliable way of rating water quality by means of a <u>single</u> all encompassing value from one scaled series of biotic index values. Different biological parameters should probably be assessed simultaneously.

Our assessment of existing biotic indices, their perceived usefulness, and their ability to meet the five basic qualities of a pollution index (Cook 1976, noted earlier in this text) have led us to the following conclusions about attributes of various biotic indices:

1. <u>Only one indicator group is used</u>. Biotic index approaches that utilize a limited indicator base (*e.g.*, a single order, family or taxon group) appear to be of restricted value in identifying a broad spectrum of water quality conditions. This is probably related to their failure to take advantage of the heterogeneity offered by inclusion of a large element of the macroinvertebrate community. The Oligochaete indices are an example of this type of approach.
2. <u>Applied to a restricted or small locality</u>. Biotic indices developed for and based on the results obtained from a single stream study, generally are so specialized as to have little or no interpretative value beyond the conditions relevant to the research from which it was derived. By default many specific biotic indices like Beak's (Beak 1965) would have to be placed in this category, more because of the lack of acceptance and adaption by others than due to intrinsic weaknesses.
3. <u>Only presence/absence or numbers of taxa data utilized</u>. Several indices are based solely on presence and absence of certain indicator groups (*e.g.*, Trent, BMWP and Belgium indices). The ecological information about abundance is lost.
4. <u>Known sensitivity is only to nutrient enrichment</u>. All workable indices were originally formulated and used to identify organic enrichment. Indices such as CBS,

ACBS, Chutter, Hilsenhoff, BMWP and Belgium indices appear to be relatively sensitive to organic pollution but are variously affected by natural environmental stresses. Other pollutant stresses (*e.g.*, pesticide pollution) were never empirically tested.

5. Used to assess a gradient of water quality. Most indices supposedly offer a continuous assessment from unpolluted to polluted conditions. However, all vary in their ability to identify intermediate conditions. The CBS, ACBS, BMWP score, Hilsenhoff and Chutter are examples. Perhaps only discrete (but coarser) levels of water quality conditions can truly be discriminated by a biotic index alone (*e.g.*, good, poor and intermediate where the latter represents an impact in between good and poor, a transitional state, or maybe an unclear assessment which requires other assessments to help clarify the biological status of these intermediate values).
6. Relative abundance is incorporated. Most successful indices utilize a relative abundance factor in their formulation (the Belgium and Trent indices are notable exceptions).
7. Independence from sample size. Most biotic indices are more independent of sample size than "total" community assessment methods (*e.g.*, diversity indices). All biotic indices are affected by species richness and/or abundance information and thus dependent upon sample size to varying degrees. No information was available to use which would allow an evaluation of the relative sensitivity of various indices to sample size thus no specific rankings for this quality between all indices can be offered.
8. Relatively easy and cost-effective. The ease of data collection and calculation of the index value varied greatly among proposed indices. All index formulas or scoring schemes were viewed as simple. Various indices required various levels of taxonomic resolution and/or enumeration of individuals or groups. We did not consider any biotic index method as too time consuming, especially when one considers the

resource commitments and time requirements necessary to chemically and physically quantify stream conditions. The documentation of the biological conditions associated with water bodies is often the most important element in the final characterization of existing water quality conditions.

9. Validity of indicator species to reveal water quality. Potentially the most important factor determining the usefulness of a biotic index in identifying the biological changes brought about by pollutant stress is linked to the validity of the assigned tolerance values. The tolerance or quality values assigned to organisms used in an index scheme must be correct and founded on scientific data and/or judgment. This selection must be made *a priori* to the application of a biotic index which utilizes the tolerance values for any specific site. The ultimate assigned tolerance value should be based on the organism's perceived or known sensitivity to a pollutant under regional habitat and water quality conditions. In practice, those indices founded on tolerance values derived from large empirical databases appear to work best.

A BIOTIC INDEX FOR KANSAS

Requirements for a Kansas Biotic Index

Clearly, there exists no "ideal" biotic index as there is no single ecological measure that in and of itself reveals all answers to all questions regarding impact of man or his activities on lotic ecosystems. However, our review and others (*e.g.*, Balloch *et al*., 1976; Hellawell 1978; Washington 1984) suggest that several indices perform quite well when confined to the geographical or habitat limits established (or inferred) for each index. In addition, some indices appear to function effectively over a broad range of environmental stream conditions indicating a greater potential for adaption to other geographic areas.

Based on the information obtained from this review, other published work and our own experience, we are proposing a biotic index system to test for use in Kansas. This proposed indicator species approach utilizing calculated biotic index values is based on the following qualities or factors that are thought to contribute to a successful biotic index approach. It is hoped that such an index scheme will help in identifying biological change brought about by man-induced alterations in the quality of Kansas streams. In part, these qualities or properties incorporate the concerns of Cook (1976).

1. <u>Identify degrees or levels of impact</u>. All of the reviewed indices apparently afford some measure of change from unpolluted to polluted conditions (at least within the conditions identified in those studies from which indices were developed). As previously mentioned all vary in their ability to identify intermediate conditions. Only the CBS, ACBS, BMWP, Hilsenhoff and Belgium indices were perceived as capable of offering a continuous assessment through an established range of values. Only the Chandler indices are open-ended in that unpolluted streams may display score values that can differ as much as a thousand or more.

Certainly the potential sensitivity of an index needs to have a broad base with respect to detecting many degrees of particular pollutant impacts. Utilizing a limited taxa base (*e.g.*, Oligochaete indices) lessens sensitivity to identify many levels of pollutant induced stresses. Such an index fails to take advantage of the heterogeneity associated with the large macroinvertebrate community (predominantly composed of insects) common in most streams. Those indices that attempt to incorporate all available indicator information from a wide variety of taxonomic groups theoretically should be more sensitive. Thus, we suggest including all insect taxa from each invertebrate collection, provided relative tolerance information is available for each taxon.

2. Limited variability due to nonpollution stress and habitat. Most references to an index's response (or nonresponse) to environmental stresses refers to those natural stresses often associated with headwater sites (*e.g.*, Balloch *et al*. 1976; Murphy 1978). Most indices appear negatively affected by such factors as temperature, altitude, water velocity, water permanence, and substratum. In general, most of these factors are associated with stream size and type. The effects of faunal changes commonly associated with natural stream succession were never specifically addressed in any of the index literature reviewed. In summary, the reviewed indices may be placed in one of several categories according to their responses to habitat and/or natural stress factors: 1) those highly influenced by factors other than pollution found in headwater streams (*e.g.*, Trent, CBS); 2) indices that are relatively unaffected by the physical properties of habitat (*e.g.*, substratum, temperature). Our literature review suggested that the ACBS, Belgium and possibly the BMWP score were minimally affected by differing environmental factors associated with various stream types or habitats; and 3) habitat specific indices that were developed for use in a very restrictive set of stream conditions (*e.g.*, riffles in permanent streams). The Beak, Chutter and Hilsenhoff biotic indices are examples of very restricted index schemes. Some of the indices may prove adaptable to broader stream conditions and still remain of value in the assessment of water quality conditions. While originally developed for use only in riffles, both the Chandler (CBS) and Trent indices have been used successful with samples taken from "pools" (Solbe 1977). Solbe's data revealed that the spatial pattern of the "riffle" and "pool values" of these two indices were very similar with "pool" scores being consistently lower

throughout the range of measured stream conditions. Balloch *et al*. (1976) also found the CBS to give comparable results, when associated erosional or depositional areas were tested.

3. Independent of sample size. We concur with Hellawell (1986) that if an index is derived from relative abundance for each of its organisms, the result becomes less dependent upon sample size. This approach has been successfully used by Chutter (1972) the average Chandler score (ACBS) (Balloch *et al*. 1976; Cook 1976) and later Hilsenhoff (1977, 1978, 1987). While not discussed it is clear that sample size may affect the results of those indices that are based solely on presence/absence information or numbers of taxa.

4. Identify impacts of various pollutants. All but Beak's river index (Beak 1965) were developed for use in assessing the biological impact of organic enrichment in lotic environments. Indicator organisms used in these indices were selected for their known sensitivity or tolerance to organic pollution but because indicator organisms are not equally sensitive to all types of pollution (Slooff, 1983), indices based on these values may prove to be very ineffective in assessing other types of pollution (*e.g.*, heavy metal, pesticide pollutants). Furthermore, one type of pollutant may or may not affect changes in aquatic communities similar to the way that changes are effected by other pollutant types.

Generally, pollutant groups or types (*e.g.*, heavy metal, sedimentation, organic enrichment) can be termed selective or nonselective, in reference to the kind of impact they impart on an aquatic community. Selective pollution would cause a selective elimination of sensitive (intolerant) species and often concurrent enhancement (increase in numbers and/or species) of insensitive (tolerant) species. Most biotic indices will document this type of alteration of the macroinvertebrate community.

The introduction of toxicants to aquatic systems represent a nonselective impact which often results in nonselective reduction in the population densities of all species with the loss of some species. The most important effect of nonselective pollutants, apart from reducing population densities and species richness, is to increase the equitability (distribution of individuals among species) of surviving species (Kovalak 1981). The impact of these concurrent changes on the assessment value of particular biotic indices is unclear.

A single biotic index approach might be successfully used to assess biological changes resulting from different selective and nonselective pollutant groups if appropriate and meaningful tolerance values could be determined for specific taxon responses to each pollutant (or type of pollutant). We have developed specific sets of tolerance values for six selected pollutant categories to be used in a biotic index. We are encouraged in this endeavor by the results of the study by Solbe (Solbe 1977) on Willow Brook (Northamptonshire, UK). Solbe found that both the Chandler (CBS) and Trent indices successfully assessed the spatial impact of zinc pollution on stream invertebrates. However, high ammonia values were also associated with the effluent. Hellawell (1977a) also noted that systems such as the saprobic system and the Trent index also respond to other pollutants but warned about their obvious limitations in this respect.

It is possible that different biotic indices may be needed to identify different pollutant types. For simplicity, we chose to begin by proposing only one biotic index scheme be used for six pollutant categories (although tolerance assessments are made independently for six pollutant categories). We suggest that this be tested across pollutant categories to determine whether or not using different biotic index schemes would be more appropriate.

5. Underlying ecological information. It is important that we consider the various biotic indices by comparing their validity in terms of the ecological information upon which they are formulated. Hawkes (1977) summarized the basic ecological changes indicative of anthropogenic water quality changes (Table 8) noting that earlier indices were essentially autecological. They utilized only the observed response of individual taxa (A of Table 8). Although this type of information was retained in later methods, it was often supplemented by synecological responses (B-E of Table 8).

He suggests that the more responses utilized in calculating the index the more sensitive the system is likely to be. Hellawell notes that the Trent biotic index incorporates only responses A and B while the Chandler score is formulated on A, B, and C responses and studies comparing these two indices consistently indicate that the Chandler method is more sensitive. The Belgium index (sensitive to only A and B responses) is reported (DePauw and Vanhooren 1983) to work well in Belgium streams. No comparative studies were available to determine if including more responses parameters (*e.g.*, C or D) would enhance its sensitivity to more specific levels of impact. If we consider all of the above indices reviewed, only the average Chandler score (ACBS), Chutter's index and Hilsenhoff's index utilize information based on responses A, B, C, and D. None of the indices covered in this report were thought to be based, in part or as a whole, on any information associated with E and F responses.

Proposed Kansas Biotic Index (Chutter-Hilsenhoff Biotic Index)

It is evident that of the biotic indices evaluated only the Chandler, Chutter and Hilsenhoff systems appear capable of incorporating all or most of the desirable characteristics needed to formulate a sensitive index that might be usable in a variety of stream conditions. It is important to note that indices such as the Belgium index have worked well for those that employ them, however, their success no doubt rest solely on tolerance values selected. Based on the available literature, the Chandler score, especially the ACBS, represents the most reliable, versatile and sensitive biotic index in general use today. The Chutter and Hilsenhoff indices could not be directly compared to the Chandler scores but apparently work very well within the regions for which they were developed. Theoretically the Chutter and Hilsenhoff indices are formulated to indicate more basic ecological information (A, B, C and D of Table 8) and in doing so tend to satisfy those qualities most desired for a biotic index.

We propose that the simpler and more mathematically flexible formulation of the above three indices be used as a basis for the Kansas index. The index formula of Chutter and Hilsenhoff (which are the same) eliminates the need for a table of values used in selecting the Chandler scores. More importantly, the former retains the use of abundance information. The primary difference between the Chandler system and the Chutter/Hilsenhoff approach is the use of abundance categories or actual sample abundances.

We are unable to assess the empirical effects of the differences between the Chandler and Chutter/Hilsenhoff systems as no studies have compared the sensitivity of the Chandler score with either the Chutter or Hilsenhoff indices under similar conditions. It may be that abundance limits are necessary to moderate the impact of abundant facultative or intermediate valued organisms on the final index value. However, the basic index formula selected for use in Kansas remains as the formula offered first by Chutter (1972):

$$\text{Chutter's Index} = \frac{\sum_{i=1}^{k}(n_i \times Q_i)}{N}$$

where, Q_i = tolerance value assigned taxa i

n_i = number of individuals of taxa i

k = total number of taxa

N = total number individuals in sample

This formula is to be used with the following sets of proposed (tentative) values derived independently for six specific pollutant categories known to occur in Kansas streams. Currently, we contend that all organisms taken from any habitat or microhabitat sampled during an established and repeatable semiquantitative timed-effort sampling methodology should be considered for inclusion in the biotic index.

HABITAT DEVELOPMENT INDEX

Introduction

Tittinzer and Kothe (1979) noted that in the use of biological indicators (especially macroinvertebrates) in assessing water quality, only sampling at hydrographically and topographically similar or equivalent points along a stream system (or between streams) will lead to comparable results which reflect water conditions objectively. Their point is well taken and most biologists recognize that to minimize the interference of abiotic habitat factors with water quality assessment, the appropriate selection of similar sampling points is critical. However, this is not always possible (*e.g.*, need to sample below effluent discharge regardless of habitat conditions) and the biologists must account for these site (and sample) differences. This problem is an obvious one for timed-effort sampling which is designed to incorporate taxa from all available site habitats and/or microhabitats. It should also be mentioned that many assessment approaches utilize species richness information but differences in richness may be related to habitat or water quality, or both.

Often habitat differences and their influence on assessment interpretations can be overcome, either by study design or by interpretive power of the methods employed. As an example, it appears that the Belgian index will discriminate between sites where water quality is different regardless of differing habitat characteristics.

However, it is our belief that a quantifiable, standard method of reporting and characterizing the habitat that was sampled is necessary so that habitat quality and its potential effects on water quality assessments can be accounted for. In the past, biologists have relied upon verbal descriptions in an attempt to explain similarities or differences which they believed may have contributed to the assessment results.

In the following text we present the ecological basis, rationale, and method for a habitat development index that we have developed and are proposing for use in water quality assessment studies as they relate to macroinvertebrate communities.

Macroinvertebrate sampling

Macroinvertebrate sampling (especially quantitative efforts) in most rivers and streams has generally been restricted to relatively shallow riffle areas which are accessible by wading and which are often regarded as relatively homogenous habitats (*e.g.*, Needham and Usinger 1956). This tendency is reflected in the development of sampling techniques (Macan 1958; Hynes 1970; Edmondson and Winberg 1971; Hellawell 1978). Many of the quantitative sampling devices and their restricted application may result in sampling bias (Resh 1979; Rosenberg 1978; Elliott and Tulbett 1978). This preoccupation with shallow water erosional areas (*e.g.*, riffles) is stimulated by obvious practical constraints associated with sampling deep water zones, stream depositional areas and other more habitat specific sites (*e.g.*, submerged tree roots) that cannot be quantitatively sampled with most existing sampling devices or techniques. In addition, we are often willing to accept the general premise that erosional zones (riffles) are among other things more productive, more species rich, more representative of stream conditions, and more representative of the basic fauna of lotic waters. We do not care to argue these views, but would suggest that in many geographic areas and in a state like Kansas lotic waters vary greatly in character. In many large rivers, lowland streams and sandbottom streams erosional areas are very restricted or nonexistent.

In general, there are relatively few methods suitable for use in deeper, slower flowing reaches of streams and rivers. Sandbottom streams are seldom studied. Some assessment of the performance of deep water samplers has been undertaken (*e.g.*, Elliott and Drake 1981a, b) but there are few descriptions of the macroinvertebrate fauna of pools available in the literature. Too often the quantitative or even qualitative efforts associated with the studies of river pools or other non-erosional zones is limited to the use of samplers such as grabs that by nature are restricted to areas where fine sediments accumulate. Such limitations in sampling strategy continues despite our knowledge that in fine substratum species and biomass are generally poor (*e.g.* Hynes 1970). The major fauna of these rivers and streams are concentrated or restricted to specialized habitats (*e.g.* debris dams, submerged logs, or cutbanks) as exemplified by the findings of Mikulski (1961).

Many workers have turned to some type of semiquantative or qualitative methods to estimate macroinvertebrate community structure in routine or surveillance studies. One of these methods is the kick method (*e.g.*, Hynes 1970; Frost *et al*. 1972) or some form of timed-effort procedure aimed at sampling available macro- and microhabitats in relation to their occurrence or importance in regard to the study objectives. These methods allow the sampling of various stream types, despite the general objections concerning attempts to compare hydrographically and topographically different streams and rivers and the apparent interpretation problems encountered with samples collected semi-quantitatively across stream types.

We propose the use of an abiotic index in Kansas on all types of streams to facilitate the use of timed-effort methods for sampling macroinvertebrates. The single largest potential variable associated with these methods is that each sample is assumed to represent a composite of potentially all available habitats sampled by the biologist. Differences in the habitat sampled can be quite large and the resulting faunal sample may reflect either habitat quality, water quality or both.

Habitat diversity

Complex or heterogeneous lotic environments with a variety of physical features inherently provide microhabitats for macroinvertebrates. Such environments generally show higher species diversity than do more simple ones (Hall *et al*. 1970; Harman 1972, Abele 1974). Jenkins *et al*. (1984) also found that the most taxa were recorded from river sites with the greatest number of habitats. Thus a sampling technique that attempts to sample all available habitats (micro- and macrohabitats) will theoretically result in sampling communities of the greatest richness or diversity present. In addition, we are often confronted with the generalized attitude that the fauna of riffles and pools are quite distinct and that samples comprised of only one or the other will also be distinct. However, this assumption may not be entirely true at least for upland stream types. Logan and Brooker (1983) examined the differences in faunas of riffles and pools from a number of studies (nine from North America and eight from United Kingdom) conducted in upland areas. Overall the number and representation of taxa in the two habitats was similar although some organisms (*e.g.*, *Simulium* for riffles; Corixidae for pools) may characterize each habitat. Some differences were noted: 1) total densities were greater in riffles; relative abundances of orders were variable; 2) only Ephemeroptera showed significant differences in density between habitats; and 3) overall there were no major differences in computed community parameters (*e.g.*, Shannon-Wiener diversity index and Jaccard coefficient) for riffle and pool faunas. This lack of strong associations between the fauna and specific macrohabitats in rivers was also noted by Jenkins and coworkers (1984).

In upland streams, the high frequency and proximation of riffle/pool sequences probably contributes to faunal similarities between the riffles and pools. In lowland reaches of streams, riffles are very restricted and more discrete, thus, greater differences between faunas may occur. Sandbottom streams, characterized by lack of distinct pool and riffle areas, will have less diverse but a different fauna than other stream types. The faunas contained in a sample will be maximized by using time-effort methods that call for sampling all available habitats whatever the stream type. Often these habitat differences are ignored or addressed in only a verbal manner by the biologist when habitat differences are thought to affect interpretation of water quality differences between samples.

It is our opinion that because biotic indices, in general, are derived from the information associated with each species sampled and its associated tolerance value, samples should include taxa representative of all stream macro- and microhabitats. Samples that are based on collection procedures which include many species should more accurately reflect the overall water quality at a particular site. However, increased sampling efforts among many habitats and comparisons among different stream types will enhance potential habitat effects upon the final taxonomic composition of a sample and biotic index values. Thus, we propose initiating the use of a scoring system for quantifying the variety of habitats available that are conducive to colonization by macroinvertebrates for each site sampled.

Proposed Habitat Development Index (HDI)

The Habitat Development Index (HDI) presented here is an assessment of stream habitat complexity which in many cases will relate to aquatic insect richness and diversity. The HDI is used to quantitatively describe the stream habitat(s) from which an aquatic insect community is sampled in a timed-effort method that includes sampling a variety of habitats. The presence or absence and relative abundance of various macro- and microhabitats are considered primary factors influencing the types of insects which inhabit a stream. We often want to distinguish between naturally occurring species compositions and pollution-induced differences in species compositions. Based on our previous discussion we must assume that streams with similar water quality may have very different insect communities if available habitats in the streams are strikingly different. Without information on the types of habitats that occur at a stream site, the contribution of habitat to the insect community composition remains a potentially unknown variance factor. Therefore, habitat differences must be considered when offering interpretations regarding biotic index or other community analyses for water quality at individual sites or streams.

Values of a quantified habitat development index are many. An HDI can help an experimenter organize stream sampling sites according to habitat similarity. This is important for studies where differences in insect community structure are to be associated with water quality parameters or other factors. An HDI can be used to help understand discrepancies found between biotic index values and associated known pollutants. For example, significant differences among HDI values may explain dissimilar biotic index values found between streams with similar water quality. Lastly, a HDI, (however limited or general it may be in structure) is helpful in converting this otherwise descriptive type of data into a standardized and repeatable form or score.

We propose the development and use of a habitat development index to be used when a stream is being evaluated for ecosystem perturbation(s) which have a potential of affecting an aquatic insect fauna directly or indirectly. The following is a presentation of the HDI which we are proposing. At this time the HDI remains untested and its quantified relationship with biotic indices and other data analysis methods has not been established. Its strongest (and perhaps weakest) attribute is its simplicity and ease of use, which we hope will encourage its use and refinement in general assessment programs.

The HDI is calculated when stream insects are being collected. Timed qualitative sampling methods for stream insects potentially allow collecting in many different microhabitats within a stream. All microhabitats which are present should be sampled at every collecting site if time allows. Sampling effort from each microhabitat can be in proportion to the availability of each type of microhabitat; or on the success rate of sampling efforts or objectives of the study. The HDI may help minimize collecting biases by offering the collector a standardized set of habitat characteristics.

Major habitat qualifiers are scored prior to sampling or as they are sampled. Habitats and habitat qualifiers include: the presence of pools, riffles and runs; average water depths of the pools, riffles and runs; riffle substrate composition; organic detritus and debris; algal masses; macrophytes; and bank vegetation. Characteristics of the habitats sampled are scored in relationship to their potential influence on habitat richness. No attempt was made to incorporate the relative or perceived value of each qualifier in relation to another. Each habitat category is defined, justified and scored as described below and the HDI compiled from a standard form (Table 9).

Riffles, pools and runs are considered as the three possible macrohabitats which comprise the total habitat for a “stream insect community”. Before commencing a timed qualitative sampling of insect fauna the collector should make a cursory assessment of the prevalence of these three macrohabitats at the stream site. Then the general availability of different microhabitats within each macrohabitat should be noted. From this the collector will decide how to partition sampling effort according to the relative availability of each macro- and microhabitat or the objectives of the study.

Minimum macrohabitat score. At the start of collection each type of macrohabitat is scored with a three if it is present or with a zero if absent. This score is placed in the right-hand column under the appropriate macrohabitat category (*i.e.*, riffle, pool or run). These values represent the minimum scores possible for any macrohabitat that will be sampled. Normal stream runs in Kansas may be loosely defined as stream areas of consistent, unbroken depth and flow, while pools are areas where deeper water occurs and water depth differs dramatically and often abruptly from adjacent stream depth. Riffle areas are characterized by swift turbulent water and uneven bottom substrates. More complete definitions of riffles and pools may be found in any number of publications dealing with the ecology of streams (*e.g.* Hynes 1970).

The scores for microhabitats sampled within each macrohabitat category can only increase the minimum macrohabitat values yielding total macrohabitat scores for riffles, pools and runs. These macrohabitat scores get larger as the microhabitats sampled within each increase in complexity. If no insects are found in a microhabitat that is sampled, the score for that microhabitat should still remain as it was evaluated.

Many of the qualifiers used in this habitat scoring system relate to the physical nature of the substrate and substrate compositions. The relationships that exist between stream insects and substrates have been well documented (*e.g.*, Hynes 1970; Minshall and Minshall 1977; Rabeni and Minshall 1977; Reice 1980) and many generalities have been incorporated in our approach. Other habitat attributes and their potential importance in contributing to the species richness of a sample is based on our own findings in Kansas. It is not our opinion that sample richness is simply and distinctly relatable to sampled macro- or microhabitats provided water quality constraints are similar, but that, in general, species richness is linked directly with habitat richness. It may well be that some qualifiers are redundant and HDI scores may vary within specific bounds (as yet unquantified) without affecting species or sample richness.

The following habitat characteristics or qualifiers are considered important in determining the relative habitat richness of a site from which a sample is obtained. Their importance and scores is a reflection of our own experiences and those of other researchers. It is assumed that sampling and thus HDI ratings will be conducted in streams under normal flow conditions. Sampling and scoring sites under extreme flow conditions (drought or high water) is unacceptable for general surveillance purposes.

<u>Average depths</u>. Water depth should be measured in each riffle, pool and run that is sampled. Average depth for each macrohabitat is then estimated and scored as 0, 1 or 2 corresponding to the most appropriate depth range indicated on the HDI form. Gore (1978) and others have identified riffle depth as an important determinant of high faunal diversity. Pool and run depth indicate water permanency and afford a measure of the refugia offered insects under various flow conditions. The fauna of intermittent streams is generally more restricted and less diverse than in permanent streams (Hynes 1970).

<u>Riffle substrate score</u>. As indicated, this score is given for riffles only and is based on the presence of boulders and bedrock, relative amounts of cobble sized substrate particles, and degree of cobble embeddedness. In general it can be stated that the larger stones, and thus the more complex the riffle substratum, the more diverse is the invertebrate fauna (Hynes 1970). Minshall's (1984) review of aquatic insect substratum relationships listed the following generalizations on substrate composition and size: 1) aquatic plants support higher densities of animals than do mineral substrates; 2) larger inorganic substrates are more productive than small-sized ones; and 3) preferences for a given substratum differ among insect species. He was quick to point out that there are many qualifiers to these generalities but indicated that intermediate sized materials maintained the highest densities.

It has been our observation that the presence of cobble-sized material can be used to judge potential insect diversity and density as it not only reflects a favored particle size for many insect species but indicates a high degree of substrate heterogeneity (Hynes 1970; Reice 1974; Osman 1978). Substrate heterogeneity provides more kinds of living places and therefore can support a greater variety of insects than a simple one (*e.g.*, Sprules 1947; Hynes 1970; Tolkamp 1980).

Cobble is defined according to Wentworth's (1922) substrate particle size classification system as being between about 6 and 26 cm in diameter. Boulders are anything greater than 26 cm. Percent cobble is scored according to the percent of riffle substrate which is cobble-sized. A single score of 0, 1, 2 or 3 for 0-10%, 11-25%, 26-50% or >50%, respectively, is given which best represents the abundance of cobble in all riffle areas sampled. We recommend the use of diagrams for estimating compositional percentages (Figure 1) as found in Compton (1962). If there is ≤10% cobble but boulders and/or a substrate of exposed bedrock is present a score of one rather than zero should be given and put in box A. A score of one for the presence of boulders and/or bedrock is justified on the basis of their value in providing suitable colonization sites which are extremely stable and add to the heterogeneity of a site.

Embeddedness is measured as the percent of vertically cross-sectioned area of cobble-sized particles that lies beneath fine sediments (< 5 mm in diameter) on the riffle bottom. Platts *et al*. (1983) first used the term embeddedness to rate the degree that larger channel or riffle particles (boulder, rubble, or gravel) were surrounded or covered by fine sediments. They initiated use of a five point rating system where the rating was a measure of how much of the total surface area of the larger size particles were covered by fine sediments (≤ 4.71 mm in diameter). In practice, the use of cross-sectioned estimates of embeddedness were simple to obtain and closely related to the actual estimated surface area that was found to be embedded. Inspection of six or more cobble-sized rocks from the sampling area usually provides a reasonable estimate.

Often the embedded portion of the cobble is distinct due to the lack of periphyton growth or color differences resulting from conditions associated with this fine sediment environment.

We have chosen to indicate the degree to which typical riffle materials (*e.g.*, gravel and cobble) may become embedded by fine sediments by estimating the percent of embeddedness of most surface occurring cobble (Figure 2). Increased embeddedness of cobble is viewed as a condition which negatively impacts substrate complexity by reducing and/or removing interstitial areas and by reducing habitat surface area. This loss or reduction in heterogeneity can result in reduced invertebrate densities and taxa richness (*e.g.* Hynes 1970).

No account is made to differentiate between the quantity and/or quality of epilithon which is associated with substrate surfaces. This characterization would be difficult to quantify in the field and according to Williams and Moore (1985) it did not significantly influence the numbers and diversity of invertebrates that colonized their "artificial" stones.

<u>Organic detritus and debris</u>. The organic detritus and debris score is based on a description of the combined material sampled within each macrohabitat type. The importance of these qualifiers has been reviewed by Minshall (1984). Organic detritus includes material such as seeds, pods, leaves, and small bark, twig and leaf fragments. These may accumulate into piles or packs. Organic debris includes larger diameter sticks, bark and logs. Four levels corresponding to increasing amounts of organic detritus and debris yield increasing scores of 0, 1, 2, and 3. The level chosen should be that which best exemplifies the composition and variety of the total detritus and debris sampled within each macrohabitat.

<u>Algal masses</u>. The importance of macrophytic algae and macrophytes (aquatic vascular plants) in providing specific habitats for macroinvertebrates has been taken primarily from the work of Percival and Whitehead (1929), Lillehammer (1966), Minckley (1963), Egglishaw (1969) and the discussions found in Hynes (1970). If algal masses are large enough to provide habitat and not just food for insect fauna, they should be sampled and scored. Algal masses consist of filamentous algal growths which may appear as small "pillows" or "beards" attached to substrate particles or as large algal beds. Thick mats of diatoms may also cover many substrates and provide habitat. Algal masses are scored only for their absence or presence (0 or 1, respectively) within each macrohabitat category when they are sampled.

Macrophytes. A score for presence and abundance of macrophytes is given for each macrohabitat category. Macrophytes include any floating-leaved, emergent or submersed types of aquatic plants. Examples are watercress, *Sagittaria*, cattails, *Potamogeton*, *Myriophyllum*, and submersed mosses. Scores of 0, 1 or 2 are given for increasing amounts of macrophytes present and sampled for aquatic insects.

Bank vegetation. A score for availability of bank vegetation as microhabitat for aquatic insects is given for each macrohabitat category. Bank vegetation can be sampled for aquatic insect fauna when any portions of terrestrial plants are submerged or exposed (*e.g.*, tree roots) under water. This will include plants growing at the water's edge as well as overhanging tree branches which dip down into the water. Possible scores are 0, 1 or 2 for increasing amounts and/or diversity of bank vegetation present and sampled.

It is our thought that bank vegetation (in and of itself) may be no different in terms of a substrate than some of the other qualifiers used (*e.g.*, macrophytes) but it is their fairly consistent occurrence at the edge of streams that makes them unique. Many streams in the central plains region have unstable, shifting sandy stream beds and/or are characterized by reduced water clarity. These features often limit or exclude aquatic vegetation. However, submerged terrestrial vegetation along a stream is highly utilized by aquatic insects to maintain populations that might otherwise be associated with aquatic plant forms. In addition, some insects are most frequently found inhabiting submerged tree roots and other microhabitats resulting from terrestrial vegetation. Jenkins *et al*. (1984) found "rare" taxa were most frequently collected from tree roots and marginal vegetation (*e.g.*, *Ranunculus*).

Calculation of the HDI

The HDI value should be calculated when sampling for insects has been completed at a single stream site. However, the microhabitat scores should be adjusted, if necessary, to omit microhabitat scores that were present and scored but where samples were not taken (*e.g.*, from lack of time). Remember that microhabitat scores should represent the actual microhabitats examined for insect fauna during the collection period whether or not any insects were collected. For each macrohabitat category that was sampled and thus received a minimum score of 3, there should be scores recorded for every microhabitat component under that category in the right hand column on the HDI form. Macrohabitat categories that were not sampled will remain as blanks in the right hand column and are considered as zero. The scores for each microhabitat should then be added within each macrohabitat category yielding a total score for riffles, pools and runs. These total macrohabitat scores are added to get the overall composite sample score. This sample score which represents the Habitat Development Index value can range from 3-40.

DATABASE FOR TOLERANCE DETERMINATIONS

Introduction to the database

Certain taxa of insects (*e.g.*, stoneflies) have long been utilized as “clean water organisms” in many saprobic and pollution index systems and their occurrence or absence is often cited as an indicator of organic pollution in numerous ecological and pollutant assessment studies and reviews (*e.g.*, Hynes 1960; Sladecek 1973; Keup, Ingram & MacKeathun 1967; Gaufin 1973a; James & Evison 1979; Persoone and DePauw 1979).

All biotic index schemes are based on the indicator species concept which utilizes the species richness associated with a water quality site and tolerance values established for each species or group of species that comprise that richness in formulating a single assessment value. While biotic indices vary somewhat we have seen that the basic approach remains very similar. Some of the most dynamic variables appear to be the tolerance scale, itself, and the actual assigned tolerance values. Authors have used rather restrictive scales (*e.g.* 0 = sensitive to 3 = tolerant) if ecological and/or toxicology data and their own experiences are limited such that only a simple, broad scale can be proposed. Conversely, if tolerance information for a variety of species is available and the researcher’s knowledge well developed, more defined scales can be established (*e.g.* 0-10 scale, when 0 = sensitive and 10 = very tolerant).

The taxonomy, distribution and general ecology of aquatic insects in Kansas is well known but specific information concerning the sensitivity of these species to various categories of pollutants occurring in Kansas streams is limited.

The establishment of meaningful tolerance values for organisms indigenous to the aquatic environments of concern is often best accomplished by examining the results of specific studies of these organisms and their observed responses to pollutants under local environmental conditions. As no comprehensive empirical database exists concerning the pollution ecology of many of the aquatic insects as it relates to specific pollutants and water quality conditions in

Kansas, other primary sources of information must be used to establish tentative tolerance values. Because of the extensive nature of estimating tolerance values we have restricted our initial biotic index to aquatic insects. Fortunately, insects are by far the most abundant and diverse group of aquatic macroinvertebrates in Kansas and are, perhaps, the best studied of all freshwater macroinvertebrates (*e.g.* Merritt and Cummins 1984; Resh and Rosenberg 1984).

We have attempted to document, in general terms, the approach we undertook in deriving the tentative tolerance values for aquatic insects proposed for use in an initial biotic index scheme. It is our feeling that while the process attempted to utilize all types of “hard” data and information in arriving at tolerance values, the procedure was, out of necessity, often modified by our subjective data interpretations and values were often “adjusted” on the basis of professional judgment and experience. We offer no apologies for this somewhat subjective approach in estimating these tolerance values but suggest that many of these “hypothesized values” be used as reference values until they can be substantiated or replaced by values derived from new findings and data.

Types of information utilized

An extensive literature search was conducted to find information on aquatic insects and their tolerances to various pollutants. Our use of tolerance refers to a species ability to readily adjust to the presence of a pollutant in its stream environment. To determine relative tolerance among insect taxa, several types of information were used. These included: ecological studies, toxicological studies, Kansas studies, tolerance values established by other researchers, identification of morphological, behavioral and physiological adaptations related to tolerance, phylogenetic relationships, geographical distributions, pollutant partitioning within streams, insect microhabitats, and personal correspondence with professionals experienced with aquatic insect ecology.

The following are the predominant resources that were identified and evaluated during the course of the establishment of tolerance values. The evaluation protocol used in screening these resources is explained in the following text.

Ecological literature

Over 200 professional publications were reviewed which contained information on insect ecology and studies on their responses to various pollutant stresses. When available the author's assessment of specific insect (species, genera or other taxonomic groups) responses to the study perturbation(s) were ranked, if possible, from 0-5. Thus insect populations that did not respond to impact might indicate tolerance (3, 4 or 5) and loss of a species or severe reduction in population numbers would indicate sensitivity (0, 1, 2). Water quality data associated with each study was examined but no attempt was made to rank or score the degree of impact by comparison with other studies' values or established water quality criteria. Instead, results of all reviewed papers were summarized and the range and mean for proposed tolerance values were calculated for each species, genus or other taxonomic group. However, our professional knowledge, experience and judgment was used to evaluate the scientific soundness of each reviewed article and thus increase the potential reliability of our estimated tolerance values. Values derived from literature sources of dubious worth were eliminated during the final evaluation and summarization process. Finally, an overall tolerance value for each taxon in each pollutant group was estimated by taking into account the mean tolerance value or the most commonly noted value derived from the selected references.

Toxicology literature

A large number of journal articles, toxicology and hazard assessment books and manuals, U.S. EPA articles and manuals, proposed or established criteria for protection of aquatic life, and other sources of both published and unpublished aquatic toxicological data were utilized in the review of pertinent toxicity tolerance information. Similar results were obtained during this literature review as with the ecological search. Few toxicity tests or bioassays have been performed on insect species indigenous to Kansas, but some information was available for North American species in genera known to occur within the state. The limited toxicological data base for North American insect species and the paucity of information on Kansas species necessitated a broader evaluation of the known toxic responses of insects to individual toxicants and/or water quality constituents. Published information containing toxicological data on any aquatic insect species was examined and used in establishing a generalized response scheme. These data were used to examine the relative sensitivity between genera or species and specific toxicants and types of toxicants (*e.g.*, organochlorides, triazine herbicides). Most of this toxicology data is being incorporated in a toxicology database for future use in the ecotoxicology program of the Kansas Biological Survey.

All toxicity data for each toxicant was compiled and taxon responses (*e.g.*, LC_{50} values) were plotted on an appropriate concentration scale. The concentration scale for each toxicant was derived from the concentrations utilized in the appropriate tests even though values often reflected concentrations that were many times greater than environmental levels associated with Kansas waters. However, placement of the responses of test species along this scale established relative sensitivities. This scale was then divided into six final concentration categories. The taxa tested with each toxicant were assigned corresponding tolerance values if they fell within a concentration grouping. This procedure was repeated for each of the toxicants for which sufficient invertebrate data was available. Toxicants or water quality constituents were arranged according to their pollutant category (*e.g.*, agricultural pesticides) and the tolerance scales for each organism versus each toxicant were collapsed into a universal scale of 0 to 5 and organisms with similar tolerances were grouped together. Concurrent with the comparison of responses of taxa to toxicants was the ranking of toxicants within a pollutant group from the most toxic to the least toxic. This was accomplished by comparing the responses of key organisms which were tested against many of the toxicants within a pollutant category. By referring to this pollutant toxic ranking, taxon responses could be adjusted to the universal scale by taking into account whether they were tested against a pollutant of limited or greater toxicity. While this method had certain limitations and was somewhat subjective, it represented a simple procedure by which organisms responses versus concentration could be plotted in a relative manner so that categories or groupings could be rated to estimate tolerance values (0-5).

Tolerance values by others

Tolerance values associated with published accounts of biotic index use and development were examined (Lewis, P.A. 1978; Jones *et al*. 1981; Rabeni *et al*. 1985; Hilsenhoff 1982,1987). Much of the current biotic index information is for indices used in Europe and Africa, therefore, species tolerance values are of limited applicability because of the ecological and faunal differences. However, many underlying principles behind the derivation of tolerance values for each biotic index were incorporated in our final decision-making process concerning the establishment of final tolerance values. This mainly applied to those values associated with organic pollution since all current biotic index schemes were developed to identify this type of impact.

Regulatory agencies or organizations responsible for the assessment, evaluation and regulation of water quality in all states were provided a mail-in survey requesting specific information on biotic indices or other methods of biological assessment (see Appendix I). We received responses to the questionnaire from 28 states. Eleven states replied that a macroinvertebrate biotic index is used by an agency in their state. Florida uses Beck's (1955) index which doesn't incorporate abundance and only counts the number of species which belong to two classes (sensitive and tolerant to organic pollution) of macroinvertebrates. New Mexico uses a biotic condition index (BCI) developed by Winget and Mangum (1979). This BCI utilizes tolerance quotients (values) that were empirically derived from abundance data for taxa found in streams characterized by alkalinity, sulfate, stream gradient (% slope), and substratum type (rubble, gravel, or sand/silt). Ohio has developed an Invertebrate Community Index (ICI) (pers. comm., Jeff DeShon, Ohio EPA). This ICI is based upon Karr's index of biological integrity that uses fish (Karr 1981). Maine is considering using the biotic index of Hilsenhoff (1982) as one of several parameters for a biological classification of streams as part of state legislation passed April 1986. Maine has not yet developed a tolerance value list for taxa and conditions in Maine.

Eight states use a macroinvertebrate index based upon the biotic index of Hilsenhoff (1977, 1982). These states are Connecticut, Illinois, Maryland, Massachusetts, Nebraska, New York, Vermont and Wisconsin. Tolerance value lists were sent to us by each of these states except New York. These values were derived by various means, for example: from Hilsenoff (1977), from other literature (although no specific citations were made), Nebraska said some of theirs originated from Kansas Department of Health and Environment (KDHE), experience in each state with monitoring organically polluted and unimpacted streams and associated macroinvertebrate fauna, and field studies. Missouri's Department of Conservation staff members provided their best estimates of tolerances for some insect taxa although their state does not currently use a macroinvertebrate biotic index (pers. comm., Richard M. Duchrow Missouri Dept. Conserv.). These tolerance lists were compiled, screened for species or genera that occur in Kansas and the tolerance values listed. This list was then used in conjunction with all other sources to determine final tolerance values for each species (taxon) to organic pollution.

Professional judgment

Throughout the selection and evaluation process the professional experience and judgment of a number of outside aquatic ecologists, entomologists, and water quality specialists was sought and their comments considered in the final selection process. In addition, the opinion of all Kansas Biological Survey staff with field experience with Kansas insects or other faunal elements, state water quality conditions or general knowledge of other professionals in Kansas concerning water quality and invertebrates was solicited and their concerns addressed. In most cases, final tolerance values are a reflection of the judgment of the local professionals whose intuition and experience allowed them to adjust the derived values or "fine tune" the evaluation system to the fauna, habitat and water quality conditions associated with Kansas streams.

Kansas and regional data bases

The Kansas Biological Survey has maintained an active aquatic macroinvertebrate inventory program in Kansas for almost 12 years. In the course of this general inventory endeavor several water quality studies were also undertaken. Most notable are those of Anderson (1979), Burkhead, Huggins and Hazel (1979), Liechti (1984) and Coler (1984). Ongoing studies by Dr. Len Ferrington and Mr. Franz Schmidt have added much to our knowledge of Kansas insects' tolerance to heavy metal pollution (Schmidt, 1986). In addition, state educational institutions, Kansas Department of Wildlife and Parks, KDHE and U.S. EPA Region VII have all conducted studies or inventories aimed at the assessment of water quality conditions and their effects on the local biota. For example, KDHE's assessment of Prairie Creek (Sedgwick Co.) has provided us with some insight into the potential impacts of volatile organic compounds on macroinvertebrates (Cringan, 1984). Unfortunately many of these reports lack sufficient taxonomic resolution or data that could be used directly in evaluating individual tolerance values. In our final selection of tolerance values we have utilized the results of all published and/or unpublished material made available to us.

All life history and ecology literature and data for those families, genera or species found in Kansas were utilized in deriving tolerance values or ratings. This proved to be a necessity because of the limited amount of information available on the normal or pollution ecology of Kansas species. It quickly became evident that we know little about the specific ecological aspects of these organisms as they exist or try to exist in polluted environments. Many generalities about some "indicator species" have found their way into the literature and, for the most part, they are accurate in their vagueness. The fact remains that we still know very little about the specific impacts of chemical pollution on insects. While often subjective in nature the tolerance values which we have presented for six pollutant categories reflects our current opinion on the tolerance or sensitivity of insects in Kansas streams.

General process used to establish tolerance values

After gathering information of any one type about as many species as possible, tentative tolerance values were given. A broad range in degree of insect tolerance was usually found. Species which displayed extremes of tolerance and sensitivity were placed at ends of the tolerance value scale. The tentative lists served as a basis for comparison to estimates of tolerance that were indicated as each type of source information was examined. We usually found that relative tolerances among species assessed from one type of information were parallel to those indicated from each other type of information. The final tolerance value list was reached after numerous comparisons of relative tolerances were made among the various sources of information.

A six point scale of tolerance values of integers from 0 to 5 according to increasing tolerance was used, where 0 = "never" tolerant; 1 = rarely tolerant; 2 = sometimes tolerant;

3 = more often tolerant than sensitive; 4 = almost always tolerant; and 5 = "always" tolerant. These ratings were selected by ranking all species found in Kansas for their likelihood of occurrence when particular pollutants are present. Values of 2, 1 and 0 were used for species which are increasingly less likely to be found in a polluted habitat. The higher values of 3, 4 and 5 were given to species which are increasingly unaffected by the presence of a pollutant.

Pollutant categories

All published information on biotic indices and corresponding organism tolerance values have been developed to assess organic pollution and other types of oxygen-demanding pollution. However, many forms of chemical pollution now occur, either singly or jointly with other pollutants. Their impact on specific organisms is only now being assessed. Toxicological data is limited to a very small fraction of the aquatic vertebrates and invertebrates known to occur in freshwater systems (*e.g.*, Pimentel 1971; Murphy 1979; Verschueren 1983; Mayer and Ellersiech 1986). Only a representative number of chemicals have been tested. Ecological studies vary widely in specific aims and methods used. Most studies relate impacts of chemicals (*e.g.*, a pesticide, a heavy metal) to effects on macroinvertebrate community structure (*e.g.*, diversity), to functions (*e.g.*, productivity rates), or to specific responses of individual species (*e.g.*, physiological effects). Often field studies limit taxonomic resolution to the generic, family or even order level and only generalities concerning overall responses can be inferred (*e.g.*, the stonefly population was reduced by increased siltation). Slooff (1983) and many other researchers have concluded that toxic pollutants are species specific and development of general tolerance values for groups of pollutants or organisms may be of little value. In addition, while the concentrations of chemical and/or physical pollution indicative parameters may be of a continuous or linear nature, biological or physiological response of species or communities may not be.

Despite the rather poorly developed data base relating to the tolerance of various organisms to toxic pollutants and the apparent lack of clear relationships between toxicants and species, we have attempted to establish a series of tolerance values for Kansas insects based on known or suspected responses to six pollutant categories. We have provided tolerance values for nearly all aquatic insects known to occur in Kansas even though in some cases, the final value for many systematic units (*e.g.*, species, genus) reflects a value based on professional judgement and inferred tolerance relationships between similar taxonomic groups.

Six categories of pollutants were chosen for which insect tolerances were to be established. These were: 1) nutrients and oxygen-demanding substances (NOD); 2) agricultural pesticides (AP); 3) heavy metals (HM); 4) persistent organic compounds (POC); 5) salinity (SA); and 6) suspended solids and sediments (SSS). For each pollutant category tolerance values for aquatic insect species present in Kansas were determined using various sources of information. Differences in the types and amounts of available information for each pollutant group varied greatly. Thus, there was some variation in the way that tolerance values could be assigned for each pollutant category.

Summarized below is a description of the procedures used to assign tolerance values within each of the six categories of pollutants. This was done for taxa in 10 orders of aquatic insects at the family, genus and species levels and was limited to those described taxa known to occur in Kansas.

Nutrients and oxygen-demanding substances (NOD)

This category includes plant nutrients (*e.g.*, inorganic N and P) and oxygen-demanding substances (*e.g.*, biodegradable dissolved organic compounds). The amount of NOD characterizes the degree of nutrient enrichment of an aquatic environment. Tolerance to NOD was evaluated by using the following information: tolerance values determined by other researchers, phylogenetic relationships, pollutant studies, insect responses in Kansas, pollutant studies, toxicological studies, and other information such as trophic habits and respiratory physiology.

Tolerance values by others: Efforts have been made in the past by other researchers to give tolerance values to many aquatic insect taxa based on their observed responses to nutrient enrichment. Most notable are tolerance assessments produced by Hilsenhoff (1977, 1982, 1987). Other tolerance lists that we used were those provided by states that responded to our survey, and the general listings found in Weber (1973), Roback (1974), Lewis, P. A. (1978) and Hellawell (1986). The tolerance values assigned to insect species found outside North America were also scrutinized in an attempt to recognize general responses that might relate to similar phylogenetic groups or species found in Kansas (*e.g.*, Woodiwiss 1964; Chandler 1970; Chutter 1972; Hellawell 1986). Tolerance values from these lists were tabularized for insect taxa occurring in Kansas. This information served as a starting point for tolerance value assignments for the NOD category. When a single taxon was assigned different tolerance values by various researchers, the most commonly occurring tolerance value was selected to best represent that taxon. After compiling this primary list, we confirmed, added and changed tolerance values for taxa found in Kansas. This was often done by using other acquired information about factors important in determining the sensitivity of aquatic insects to high levels of nutrients and low oxygen conditions. The following will outline the steps taken to attain the final list of tolerance values for the NOD category.

Phylogenetic relationships: Taxa found in Kansas but not assessed for tolerance by other researchers were given tolerance values based on taxonomic similarity (phylogenetic relatedness) to taxa with assigned values. For example, when only one species of a genus had a tolerance value, all other species in this genus would be given the same tolerance value. Assignment of a single tolerance value for a higher level taxonomic group would be based on the most common tolerance value given for taxa within that taxonomic group. More weight would be given to tolerance values of the taxa more frequently found in Kansas and less weight to rare taxa. When this was completed an entire taxonomic list from the family to the species level for taxa which occur in Kansas had been given preliminary tolerance values.

Pollutant studies: Next, information was evaluated which was found in existing ecological reports that correlated the degree of nutrient impact with the kinds of insect communities present. Many of these studies concerned sewage treatment plant effluent discharges (*e.g.*, Donald and Mutch 1980; Wynes and Wissing 1981; Kondratieff and Simmons 1982). Exemplary studies of other nutrient loading sources were acid-mine drainage (Moon and Lucostic 1979), leaf litter (Mackay and Kersey 1985) and paper pulp effluent (Rabeni *et al*. 1985). Several studies have been done in Kansas (*e.g.*, Anderson 1979, Coler 1984) and tolerance information derived from these were given more weight.

Most studies examined differences in insect community structure between sites differentially polluted by nutrients. Changes in the composition of insect communities (*e.g.*, declines in species richness) that could be related to the introduction of nutrient loads provided evidence for insect sensitivity to NOD impact. Relative abundances of insect species were compared at control sites or pre-impacted sites versus impacted sites. Insects that persisted in environments with high amounts of nutrients or low dissolved oxygen were considered to be more tolerant than insects that declined in abundance or were eliminated.

The most sensitive (intolerant) and the most tolerant insect species established the two end-points by which a scale of tolerance values was developed. Species found within each ecological study in the literature were given tolerance values scaled relative to each other as integers from 0–5 from most sensitive to least sensitive (*i.e.*, from least tolerant to most tolerant). For all taxa common to more than one study for which we assigned tolerance values, the most frequently assigned tolerance value was selected to best represent a tolerance value for this taxon. Tolerance for these taxa were then compared to the preliminary assignments that used tolerance evaluations of other researchers and phylogenetic relatedness. Adjustments were made with greater weight placed on the findings of the pollutant studies.

<u>Toxicological studies</u>: Laboratory studies which established LC_{50} values for dissolved oxygen (DO) concentrations were found for various species within several different genera of aquatic insects (*e.g.*, Nebeker 1972; Gaufin 1973b; Surber and Bessey 1974). These taxa were ordered for increasing tolerance to low DO relative to one another according to their LC_{50} values. This information was used to estimate the final tolerance values for these taxa and other taxonomically similar species.

<u>Other information</u>: Further information used for adjusting tolerance value assignments came from the identification of taxonomic specific factors, *i.e.*, characteristics of taxa that may influence the ability of insects to tolerate and thus occur in nutrient enriched environments. These included trophic habits and physiology (examples are given below). Such factors were used especially for those species for which tolerance value determination could not be made utilizing ecological data or other researchers' tolerance value lists. Again taxonomic relatedness was relied upon for determinations of tolerance values for species with otherwise limited information.

Functional feeding group classification was used to help assess tolerance for NOD. There has been found a correlation with the occurrence of NOD pollutants and the presence/absence of species belonging to particular functional feeding groups (Kondratieff *et al*. 1984; Wiederholm 1984). Often heterotrophic microbiota (bacteria, fungi, protozoa) increase under conditions of high nutrient loading resulting in a corresponding increase in insects that utilize heterotrophic microbes (Kondratieff and Simmons 1982; Kondratieff *et al*. 1984). At stream sites where nutrients were highest, collector-filterers (*e.g.*, Hydropsychidae, *Isonychia*) were most abundant, collector-gatherers (*e.g.*, Orthocladiinae, Caenidae, Ephemeridae, Ephemerellidae) were less abundant, and scrapers (*e.g.*, Baetidae, Heptageniidae) declined the most at the nutrient enriched sites and never returned to abundances found at reference sites. For insects in Kansas we assigned relative tolerance decreasing sequentially for collector-filterers, collector-gatherers and scrapers using the classification of functional feeding groups of Merritt and Cummins (1984).

Differences in respiratory mechanisms among insect taxa were considered to influence the ability of the taxa to tolerate low dissolved oxygen conditions which can occur in streams with a high influx of NOD. Merritt and Cummins (1984) describe and give examples of eight respiratory options of insects. By utilizing these examples, it was judged that insects with open tracheal systems but no gills would be more tolerant of low dissolved oxygen. Examples are air-breathing taxa such as *Eristalis*, *Psychoda* and *Culex*; taxa that get oxygen from air-storage bubbles such as Corixidae and Dytiscidae; and plant breathers like *Donacia*, Ephydridae, Culicidae and certain Syrphidae. Certain Chironomidae (*e.g.*, *Chironomus*) and some Notonectidae (*e.g.*, *Buenoa*) that possess hemoglobin were also considered tolerant to low dissolved oxygen. Conversely, insects which have closed-respiratory systems and/or gills were given lower tolerance ratings.

There was much general and some taxonomic specific information that was found in literature that discussed effects of nutrient enrichment on stream macroinvertebrates as a part of but not central to the focus of the publication (*e.g.*, Cairns and Dickson 1971; Nalepa and Quigley 1980; Hellawell 1986). This data helped us form some general concepts about how nutrients have been found to affect aquatic insects. Professional judgments made by Kansas Biological Survey scientists based on their experience with insects and stream habitats in Kansas were used in the final tolerance value adjustments.

Suspended solids and sediments

The suspended solids and sediments category includes inorganic and organic compounds that occur as particulate matter in the water and/or as settled particles on the streambed and its substrates. Information used to determine tolerance values for SSS included pollutant studies, insect morphology, physiology, microhabitat preferences, trophic habits and phylogenetic relationships.

Pollutant studies: Pollutant studies were evaluated to make our first general assessments of relative insect tolerances to suspended solids and sedimentation. The effects of SSS on stream insects have been studied as they occur with logging operations (Tebo 1955; Welch *et al*. 1977; Graynoth 1979; Newbold *et al*. 1980), quarry activities (Gammon, 1970), agricultural practices (Welch *et al*. 1977; McCafferty 1978); other forms of man-induced and natural impacts (*e.g.*, oil sand erosion, Barton and Wallace 1979a, 1979b; mine tailings, Duchrow 1978; volcanic ash, Gersich and Brusven 1982; and Brusven and Hornig 1984), and various industrial effluents (Nuttall and Bielby 1973; Hilton 1980). Apparently no Kansas studies have been conducted that examined the potential impact of SSS on stream macroinvertebrates. Often the assessment of SSS impacts from ecological field studies was complicated by the co-occurrence of other pollutants in the study areas (*e.g.*, sediments and metals, Duchrow 1983; oil sand and flooding, Barton and Wallace 1979a; sediment and organic enrichment, Lemly 1982; sediments and toxicants, Van Hassel and Wood, 1984).

Very little experimental work has been done on siltation and aquatic insects. However, the work of Brusven and Prather (1971) on a small Idaho stream and related laboratory studies proved to be very useful in establishing some tolerance values. Usually comparisons in these studies were between different locations along a stream or between different streams with different amounts of suspended solids or sedimentation problems. Insect species were considered to be sensitive if they were reported as decreasing in abundance more than other species where

SSS pollution occurred. It was usually possible to scale the tolerance responses noted in these studies.

The procedure for examining ecological data for effects of suspended solids on stream insects was initiated by relating the different degrees of SSS pollution to changes in relative abundance of insect species. When comparisons among studies were made, specific species (or taxa) always appeared among those which were least tolerant to SSS while other species were consistently tolerant. Species which were at neither extreme were given intermediate tolerance values, but these should not be interpreted to correlate with intermediate levels of SSS pollution. The loss of sensitive species and gain of tolerant species may not have a linear relationship with the concentration of suspended solids or other measures of siltation or sedimentation. It may be that small additions of SSS may be as harmful as large ones if sensitive species have a threshold response to SSS and show little decline in abundance until a particular "critical" level of SSS is reached and then their numbers drop precipitously. Factors irrespective of the specific level of SSS are probably involved in determining presence/absence of specific taxa (*e.g.*, substrate attachment).

For most insects the impact by SSS appeared to be determined more by when and for how long an SSS load was present rather than how much. A load of SSS can be distributed throughout a stream in many different ways and influenced by such factors as flow and bottom contour (Brusven and Prather 1971; Gammon 1970; Lenat 1983). The composition and structure of the bottom substrate and the availability of suitable refugia from an SSS load are features of a stream which will affect the composition of aquatic insect fauna (Brusven and Prather 1971; Gammon 1970; Nuttall and Bielby 1973; Hynes 1960; Brusven and Hornig 1984). Length of duration of sedimenting particulates will also be a determining factor in the severity of an SSS impact. Short-term SSS impact events usually result in short-term effects on stream biota when the usual bottom characteristics of the stream are quickly restored (Gammon 1970).

The level of SSS pollution can be measured in different ways and this will affect any abiotic determination of relative amounts of SSS pollutants. There may be uneven distribution of particulates among microhabitats at a stream site, different types of sedimenting particles, and temporal variability in deposition. Thus, an exact measure of the amount of SSS pollution can be difficult (Gammon 1970; Baker 1984). All such factors confound attempts to compare the degree of SSS impact in one study with that in another study.

Toxicological studies: There were no "toxicological" studies used in determining the tolerance values for the SSS category. A study by Brusven and Hornig (1984) which experimentally controlled additions of sedimenting volcanic ash to insect taxa in laboratory conditions found no toxic effects on 10 aquatic species of Ephemeroptera, Plecoptera, and Trichoptera. In other toxicological studies sediment effects were not isolated since sediments which were added contained other directly toxic substances such as heavy metals or certain organic chemicals.

Other information: Insect tolerance to SSS input appears to be related to various morphological, behavioral and physiological adaptations. These include respiratory apparati, modes of feeding, animal mobility, and microhabitat.

Insects which are unable to protect their breathing surfaces from silt accumulation will be less likely to function adequately under SSS impacted conditions. Gills positioned dorsally versus ventrally as in some Ephemeroptera and Plecoptera would be better suited to high SSS conditions (Hynes 1970; Roback 1974). Likewise, the elongate terminal abdominal segments of certain Gomphidae (*e.g.*, *Aphylla*) might increase tolerance to SSS. Gills protected by hairs, plates or other structures would also be advantageous (Caenidae, Baetiscidae, *Hexagenia*, and *Potamanthus*) (Merritt and Cummins 1984). Silk plugs put in the ends of cases by some Trichoptera might provide some protection. Surface breathing species (*e.g.*, *Hydrophilus*, *Culex*, and *Gyrinus*) and species which can produce hemoglobin (*e.g.*, *Chironomus* and some Notonectidae) were also considered more tolerant.

Insects may lose locomotor ability when sediments accumulate on their bodies. Species which produce portable cases would have an advantage (Nuttall and Bielby 1973; Hynes 1960; Grimas and Wiederholm 1979; Brusven and Hornig 1984; Warwick 1980). Examples include the chironomid genera, *Constempellina* and *Stempellina*, and the trichopteran genus, *Leptocerus*, as well as other cased Trichoptera. Insects with hold-fast mechanisms that require smooth substrate surfaces for attachment may suffer if these surfaces are eliminated (Hynes 1960). Sprawling and burrowing taxa (as designated in Merritt and Cummins 1984) were considered to be tolerant because they are naturally associated with stream sediments (Hynes 1970; Nuttall and Bielby 1973; Hynes 1960). Many are equipped with long legs and claws better suited to loose sediment surfaces (*e.g.*, *Pseudiron*). In general, sessile species would be more sensitive than mobile forms.

Microhabitat preferences were also used in assessing SSS impact on organisms. Bottom dwelling insects considered most sensitive to SSS were those associated with small interstices or with stable substrata. In contrast, insects were considered more tolerant if they were burrowers or any type that was normally associated with shifting sediments such as sand and silt. Other insects thought to be less affected by SSS were those that swim in the water, climb or cling to plants, or occur with the neuston.

In addition, trophic habits or feeding modes were considered. Filter-feeding mechanisms may be hindered by high concentrations of particulates and/or food quality may be reduced. Trichopterans which build silk nets are adversely affected (Gammon 1970; Hynes 1960) and need to spend more energy to keep nets free of SSS. Grazers of periphyton were considered more sensitive since primary production may be affected by reduced light conditions in turbid waters (Hynes 1960). Food quality for scrapers and collectors might also be reduced. Predators that rely on visual cues to find prey (*e.g.*, some Odonata and Plecoptera) could also be adversely affected. SSS can also harbor large populations of fungi (Hynes 1960) and bacteria (Lemly 1982) which can be infectious for some taxa.

Considerations of these morphological, behavioral and physiological adaptations, most of which were taken from Merritt and Cummins (1984), were tallied for the corresponding taxa. These were used to make judgments about tendencies towards sensitivity or tolerance to high levels of SSS. Tolerance values were given to all Kansas taxa by supplementing the tolerance assessments made from the pollutant studies with the insect life history information and phylogenetic relatedness.

Tolerance values for the SSS pollutant category may not reflect the presence of SSS pollution in the same manner as the NOD category. It may be that high sediment loading and periodic introductions of fine inorganic materials into many Kansas streams is a natural phenomenon resulting more from geology and land form than from the activities of man. However, changing land use and other direct human activities have probably influenced the frequency, duration and intensity of SSS pollution in some river basins. The ability to relate "background" natural SSS from introduced SSS pollution (either chemically, physically or biologically) may be difficult without additional research.

Salinity

Salinity was considered as a "water quality parameter" that directly affects presence and absence of specific taxa of aquatic biota. Salinity as used herein refers to dissolved acids, bases and salts often measured as conductance or salinity (chloride concentration). Originally we attempted to address "dissolved solids" as a pollution category but was abandoned in favor of a salinity category. This was done to avoid confusion with the use of the commonly used measures for dissolved solids by means of oven dry weighing of filterable "solids" (APHA Standard Methods 1985). We found no literature that directly or indirectly correlated sensitivities of aquatic insect fauna to this latter measure of dissolved solids. In contrast, there are known physiological adaptations that aquatic organisms must have to salinity and a single major review paper was available that documented tolerances of some insects to saline environments.

The following types of information were utilized in estimating tolerance values for SA: ecological field studies, physiological adaptations, professional judgment, and phylogenetic relationships.

<u>Ecological studies</u>: Ecological studies about effects of high salinity on macroinvertebrate insects were scarce. Presence/absence data from the studies that we found (*e.g.*, Canton and

Ward 1981) indicated differences in tolerance to high salinity among insect taxa. An extensive database was compiled by Roback (1974) which included ranges of chloride concentrations found with many different aquatic insect taxa collected throughout North America. Collection sites were generally associated with aquatic systems influenced by industrial and municipal wastes. These data were used to arrange many taxa sequentially according to chloride concentrations in which they were found. Kansas taxa mentioned in these studies were arranged according to their relative tolerances. The SA category had the least amount of ecological literature in comparison with other pollutant categories and we relied primarily on the synopsis given by Roback (1974).

Physiological adaptations: Specific information regarding the physiological, behavioral or morphological adaptations of some taxa for tolerating high salinity were also scarce. Certain species are able to osmoregulate in higher salt concentrations than other species and were considered more tolerant. Osmoregulation by absorption is more important than control by excretion for most insect species that live in waters containing little dissolved Na, K or Mg ions (Kapoor 1978, 1979). It has been shown that some aquatic insects exhibit decreasing chloride uptake with increasing salinity (Wichard 1976; Wichard, *et al*. 1975). This is advantageous when salts concentrate in temporary pools. Many insects seem unable to excrete salts at a sufficient rate to compensate for high saline conditions and thus should be more sensitive. However, we did not have information on specific taxa that would be sensitive.

Professional judgments of insect distributions in Kansas: Distributions of aquatic insect species between streams in Kansas provided some indication of insect tolerance to salinity. We also attempted to relate insect species distributions to local geology and the likelihood of geological contributions to salinity or specific conductance in streams. Judgments were made about tolerance and sensitivity based on presence/absence data from the general collections of the Kansas Biological Survey. These were added to the relative tolerance value list and conversions were made to the six point scale.

The tolerance value list for all Kansas taxa was completed by using phylogenetic relationships for all species for which no other information was available.

Heavy Metals (HM)

Heavy metals included all the alkaline earth elements with atomic weights greater than Calcium. The tolerance values for the heavy metals category were determined by considering toxicity values from laboratory studies, pollutant studies, Kansas studies, insect natural histories and heavy metal partitioning in stream environments.

Toxicological studies: Acute toxicity tests with heavy metals may be useful in indicating relative sensitivities between aquatic insects and metal exposures, but these tests have little environmental meaning since heavy metals are rarely found in LC_{50} concentrations even in highly polluted waters (Clubb *et al*. 1975; Warnick and Bell 1969; Rehwoldt *et al*. 1973). Nonetheless, toxicity values were compared between insect taxa and provided a means of establishing relative tolerances. Direct comparisons between long-term chronic tests (*e.g.*, EC_{50} values) and the short-term lethality tests (*e.g.*, LC_{50} values) were avoided due to the differing levels of sensitivity associated with each type of test.

Various chronic responses of aquatic insects have been measured at concentrations occurring in the environment. Heavy metals have been shown to affect molting and emergence in long-term, chronic toxicity tests at concentrations much lower than levels associated with acute tests for lethality (Clubb *et al*. 1975). Larvae of the mayfly, *Ephemerella ignita*, were slower to develop and exhibited a reduced emergence rate after exposure to only 5.2ug/L Cobalt (Sodergren 1976). *Chironomus tentans* was similarly affected by Chromium, Zinc and especially Cadmium that was bound to sediments (Wentsel *et al*. 1977, 1978). It has also been noted that net-spinning capabilities of hydropsychid caddisflies was affected by high copper concentrations as well as other heavy metals (Besch *et al*. 1979; Petersen and Petersen 1983). Taxa for which toxicity studies were found were sorted relative to each other by assuming a direct correlation of increasing toxicity values with increasing tolerance for each heavy metal. General trends among taxa for all the different HM test results were established and combined into a single set of HM tolerance values (as described earlier in the toxicology literature section).

Pollutant studies: Relative tolerances were established among taxa found in pollutant studies where heavy metals were involved by comparing the relative abundances for each taxa present at heavy metal polluted stream sites (*e.g.*, Brown 1977; Armitage 1980; Winner *et al*. 1980; Specht *et al*. 1984). The relative tolerances for taxa from the pollutant studies were combined with those from the toxicological studies. The general trends in HM tolerance in particular orders and families of insects were similar between toxicological and pollutant studies (*e.g.*, mayflies > caddisflies > midges, most tolerant). Comparisons of generic and species responses were usually not possible between studies because taxa were different. A scale of 0–5 was used at this point and a list of tentative tolerance values were created for the limited taxon base associated with this type of data.

Kansas studies: Many of the research findings of Dr. L. C. Ferrington (KBS) concerning streams of southeastern Kansas impacted by metals were used to generate tolerance values for some species. The results of heavy metal impacts on the macroinvertebrate fauna of Short Creek (Cherokee Co., KS) were also utilized in deriving specific tolerance values for various aquatic insect species (Schmidt, 1986).

Other information: Tolerance values needed to complete the list were established by considering differences in morphology, habitat and phylogenetic relatedness between taxa. Characteristics considered as beneficial to certain taxa include those which decrease the amount of contact between insect and HM. For example, the cases of certain caddisflies (*e.g.*, *Limnephilidae*) may protect the insects from contact with sediment bound HM more than non-cased insects (Brown 1977).

Heavy metals partition into various locations of stream systems much like agricultural pesticides (Figure 3). The partitioning of HM by adsorption to bottom sediments might negatively affect bottom-dwelling insect species. Cadmium has been shown to accumulate in grazers, collectors, and predators at high, intermediate and low levels, respectively (Selby *et al*.

1985). Selby and co-workers related the differences in bioaccumulation among insect taxa to greater direct contact with the metals in their microhabitats and through their food sources. Some bottom-dwellers have been used to monitor HM concentrations in aquatic systems (Nehring 1976; Nehring *et al*. 1979). Thus, all bottom-dwellers (as given by Merritt and Cummins 1984) were considered more sensitive than other species.

Differences in biomagnification of heavy metals between aquatic insects were not thought to affect presence/absence of taxa or population numbers. Biomagnification of heavy metals in insects is not known to occur (or has not been well studied) within aquatic insect communities. Insects at a higher trophic level such as predatory insects were not considered more sensitive.

Phylogenetic relationships between taxa was used to give tolerance values to those taxa for which specific tolerance information was unavailable.

Agricultural pesticides

The agricultural pesticides category included organic compounds and mixtures that are used as herbicides and insecticides, both persistent and rapidly degrading types. Insect tolerances to AP were based on toxicological data, pollutant studies, insect natural history, pesticide dynamics in streams, and phylogenetic relationships.

Toxicological studies: Toxicity values (LC_{50} and EC_{50}) were compiled from the literature for many different pesticides. The values for each species and a specific pesticide were arranged in sequential order. This produced relative tolerances among various taxa for each pesticide. If toxicological tests differed markedly in duration (*e.g.*, 30 days versus 24 hours), toxicity values were not directly comparable and interpretations were modified accordingly. Ideally, toxicity values should be determined under the same experimental conditions (*i.e.*, temperature, pH, DO) to make comparison more meaningful, however, this was seldom the case. Subjective interpretations of the relative toxicity of various pesticides (and all other toxicants) to different insects was often necessary. Tolerance assessments for a pesticide were considered most reliable when numerous species had been tested.

After relativizing insect tolerances among insect taxa for each pesticide, we compared and combined the assessments into one broad range of relative tolerances to agricultural pesticides in general for all the species with toxicological data. Generally, the relative tolerances estimated among the various taxa for any single pesticide were similar to those generated for other pesticides. The relative tolerances were expressed on a scale of 0–5. It should be noted that tolerance values do not correspond to specific LC_{50} values.

<u>Pollutant Studies</u>: Aquatic field studies where one or more pesticides were present at the experimental sites were used to estimate insect tolerances to agricultural pesticides in environmental situations. Ecological studies were usually of the following types: microcosm studies (*e.g.*, Arthur *et al*. 1983; Yasuno *et al*. 1985); studies on natural systems with experimental applications of AP (*e.g.*, Wallace *et al*. 1973; Mulla and Darwazeh 1976; Eisele and Hartung 1976; Sebastien and Lockhart 1981); or detection of concentrations of AP resulting from AP use on adjacent land (Courtemanch and Gibbs 1980; Clements and Kawatski 1984). Books also provided various types of ecological information specifically relating to pesticides and aquatic insect fauna (*e.g.*, Brown 1978; Hynes 1960, 1970; Muirhead-Thomson 1971; Hellawell 1986). Relative tolerance was determined in the same fashion as done in other categories and described in the NOD category. Relative tolerances were then compared to those from the laboratory toxicological assessments. Resulting inconsistencies were generally resolved by giving more weight to the ecological information.

<u>Other Information</u>: Insect species which were not part of a toxicological or ecological study were given tolerance values based on natural history considerations and pesticide partitioning (Morley 1977). Feeding and microhabitat were used to predict which taxa might have more exposure to harmful amounts of AP (Duke 1977; Edwards 1977; Haque *et al*. 1977; Merritt and Cummins 1984; Wiederholm 1984).

Pesticides enter surface waters and can be found in soluble and particulate fractions of the water. They can adsorb to plants, sediments and at the air/water interface (Figure 3). Insect fauna which are associated with these primary sinks for pesticides were considered the most likely to be negatively affected especially by chronic exposures. Bottom-dwelling species were considered to be the most susceptible life forms to AP pollution. Insect life forms were considered according to increasing tolerance from burrowers, scrapers, bottom sprawlers, neuston or plankton feeders to swimmers and predators.

Life history and pesticide partitioning considerations were used to adjust taxa along a single six point tolerance scale where the focal points were based on relative tolerance derived from the ecological field studies and toxicological data. The final tolerance value list for AP was completed by filling in values for all other taxa by phylogenetic relationships.

Persistent organic compounds (POC)

Persistent organic compounds are organic compounds (including agricultural pesticides) that resist degradation and/or elimination from the environment. Those used were primarily PCB (aroclor), dieldrin, aldrin, DDT, endrin and lindane.

Tolerance values for the POC category were derived in the same way as for the AP category. Types of information used were the same: toxicological studies, pollutant ecology studies (*e.g.*, Moye and Luckman 1964; Ide 1967; Hatfield 1969; Clements and Kawatski 1984), insect life histories, patterns of pollutant partitioning in streams, and phylogenetic relationships. In general we relied heavily upon information gathered on non-persistent agricultural pesticides and applied them to the POC category.

Insect morphology and living habits that were important for the AP category were assumed to be applicable to POC. The differences between tolerance values for AP and POC were made primarily by microhabitat preferences and pollutant partitioning within streams. Insect taxa which live in the sediments were considered most likely to be sensitive to POC and this was given more weight than in the AP category, especially for taxa that were given relatively high tolerance values for AP. Scrapers and collectors (both filterers and gatherers, as identified according to Merritt and Cummins 1984) were given more tolerant ratings than the bottom-dwellers. The objective was to emphasize the sensitivities of taxa to long-term exposures of POC. The resultant tolerance values for POC were lower compared to AP tolerance values except for Ephemeroptera which were already very sensitive to AP (Figure 10).

TOLERANCE VALUES FOR KANSAS INSECTS

List of tolerance values for six pollutant categories

Appendix II contains the tentative tolerance values that we assigned for all taxa (families, genera and species) of aquatic Insecta that we know occur in the state of Kansas. The taxa are listed alphabetically within each of ten insect orders: Coleoptera, Diptera, Ephemeroptera, Hemiptera, Lepidoptera, Megaloptera, Neuroptera, Odonata, Plecoptera, and Trichoptera. The six pollutant categories are designated in this list as follows: NOD (nutrients and oxygen demanding substances); AP (agricultural pesticides); POC (persistent organic compounds); HM (heavy metals); SA (salinity); and SSS (suspended solids and sediments).

This list for the most part represents the current systematic status of most taxa, however, nomenclatorial and systematic changes within specific groups (*e.g.*, Chironomidae) are volatile and this list may already be incomplete.

Summary of our tolerance values for Kansas and comparisons to other states

There is a similar pattern in the tolerance values that we gave for aquatic insects in Kansas (for the NOD category) with those given by others for insects in other states. We selected tolerance value lists of four states from our survey, relativized values to a six point scale of 0 to 5, and calculated some descriptive statistics of the tolerance values given at the generic level in six orders of aquatic insects. Lists we used were from Illinois (Illinois Environmental Protection Agency), Massachusetts (Dept. of Environmental Quality Engineering, Division of Water Pollution Control), Vermont (Dept. of Water Resources and Environmental Engineering), and Wisconsin (Dept. of Natural Resources, as published by Hilsenhoff 1987). Four of the five states had similar tolerance values when comparing the overall means of six major insect orders (Plecoptera, Trichoptera, Ephemeroptera, Coleoptera, Odonata and Diptera) (Figure 4). The Illinois overall mean was lower than the other states including our Kansas values. This probably reflects the selective use by Illinois of a 12 point scale of tolerance values that extends only some of the most tolerant taxa towards the higher end of the tolerance scale. Similarity among some states was expected since all states have relied upon the tolerance value list of Hilsenhoff (1977, 1982) produced from many years of empirical data collected in Wisconsin streams.

Some differences among states can be seen in an examination of the frequency distributions of tolerance values along a six point 0 to 5 scale for the five states (Figure 5). The most distinct difference is that the values from Wisconsin group into two classes. This contrasts with the centrally concentrated values for Kansas, Massachusetts and Vermont and with the increased frequency towards the lower end of the scale for Illinois values. It is unknown how these differing tolerance values among states would be reflected in series of biotic indices calculated along a gradient of stream sites from low to moderate to high levels of pollution (presumably, this is pollution from nutrient loads and oxygen demanding substances).

The rank order of mean tolerance values when comparing across six major orders of aquatic insects were similar for all five states (Figure 6). Plecoptera were always most sensitive. Trichoptera and Ephemeroptera were similar to each other and second most sensitive. Coleoptera and Odonata were more variable but generally more tolerant. Diptera was always the most tolerant order. Figure 7 depicts these comparisons across states within each order. Except for Ephemeroptera, the Kansas means for each insect order were similar to three or four of the other states' means. The Ephemeroptera mean for Kansas was significantly higher (one-way ANOVA, $p=0.002$; Fisher's LSD, $p <0.05$). Ecological implications of a higher Ephemeroptera mean are not known. Over 60% of the 31 genera from Kansas were given a tolerance value of 2. The frequency distribution of tolerance values for Ephemeroptera for all five states is presented in Figure 8.

In general, tolerance values that we chose for the NOD category for Kansas insects appear to be similar to the tolerance values used for insects in four other states. This is not altogether surprising since, as explained above, 1) we initially reviewed and used some of these values for our preliminary tolerance value list for the NOD category; and 2) the other states also borrowed tolerance assessments from Hilsenhoff (1977, 1982) for Wisconsin; and 3) this tolerance pattern among insect orders is the most common pattern noted in the literature (see review by Hellawell, 1986). Still, their utility in a biotic index and accuracy for indicating streams of low, moderate and high NOD impact, remains to be tested empirically across a wide variety streams in Kansas.

Summary of tolerance values for the six pollutant categories

Not all pollutant categories yielded equivalent mean tolerance values for insects in Kansas (Figure 9). Mean tolerance values represent the average occurrence of sensitive or tolerant taxa in Kansas to the particular pollutant type. The heavy metals (HM) category mean of 1.62 was significantly lower than all other category means ($p<0.002$,ANOVA; $p\leq0.05$, LSD tests). AP, SA and SSS category means of (2.84, 2.80 and 2.88) were highest and significantly higher than NOD and POC means (2.46 and 2.44). Although the same six point integer scale from 0 to 5 for tolerance assignments was used in each category, numerically equivalent tolerance values (individual or mean values) from different categories should not be interpreted as representing absolute or "actual" biological tolerance equivalencies. Biotic indices from different pollutant categories will not be strictly comparable. An interpretative scale to indicate a relative degree of impact will not necessarily be the same from one pollutant versus another. Tolerance value assignment was done independently for each pollutant category. The relative scale with six levels of tolerance was assigned more with respect to known extremes of impact of each pollutant type in Kansas. All taxa were then placed on this six point integer scale within those extremes.

The six orders of insects have some "apparent differences" in sensitivity (*i.e.*, tolerance) to the various pollutant categories. A breakdown of the overall mean into means of the genera within six different orders of insects are presented in Figure 10. The resulting differences in mean tolerance values when compared across categories for a group of insects are difficult to interpret (for the reasons noted above). It would certainly be interesting and informative if one could determine if particular insects (or groups) are more (or less) tolerant to one type of impact than another.

The rank order of the six major insect orders mean tolerance values were not identical for each pollutant category. However one major pattern that was seen previously with the NOD category did occur (Figure 11), (*i.e.*, Plecoptera, Trichoptera and Ephemeroptera means are lower than the means of the other orders). Within these two groups the most sensitive or tolerant group (as expressed in a mean of generic tolerance values) varies.

The overall frequency distributions of tolerance values were very similar for each category (Figure 12). This is not surprising since, for each category, tolerance was assessed according to the same basic guidelines: 1) The 0 and 5 tolerance values were reserved for taxa which were considered to "indicate" extreme conditions. 2) Intermediate values of 1 and 4 were given for a status of definite sensitivity or tolerance (respectively), where our confidence was based on both quantity of data and types of information. 3) The central values of 2 and 3 were given to all the taxa which were reported to have sensitivity across a broad range of pollutant conditions (*i.e.*, facultative); or were known to be sensitive to (value 2) or unaffected by (value 3) moderate levels of a pollutant. The predominant result of following these guidelines was that for any single pollutant category ≥ 40% of the insect genera were given the same tolerance value (Figure 12). The most often assigned tolerance value was a 3 in every category except for heavy metals which was a 1. The second most frequently assigned tolerance value was given to 15-30% of the genera, and this tolerance value was usually a 2. Only 2-10% of the genera were given the extreme values of 0 or 5. Since the ecological significance of having a tolerance in the middle of the range is the least understood (or has several different interpretations), the effect on a final biotic index value is problematic. The predictive capabilities of the taxa with the middle tolerance values and a biotic index which weights these most common values may overshadow the indicator values of the taxa with more extreme tolerance values. Other types of guidelines for tolerance assignments might be better. It is possible that the discrete tolerance values should not be distributed at equal intervals between the minimum and maximum values. We suggest that the tolerance values as currently assigned be used to compare known sites of various levels of each pollutant category against unimpacted reference sites not only as a biotic index value but also to look at the frequency distribution of the tolerances that appear in the communities. The true distribution of tolerance values among taxa in a community is probably one of the most important characteristics which needs to be accurately assessed so that appropriate weighing factors for calculating a biotic index value can be made. In spite of independence in tolerance assignments between categories, the net result of using similar criteria for assigning relative tolerance was that Kansas taxa were apportioned along a six point arithmetic tolerance scale at similar frequencies for each category as depicted in Figure 12.

DISCUSSION

The primary objective of this research effort to develop a biotic index system for use in Kansas to monitor and assess biological changes relatable to water quality conditions (*e.g.*, increased organic enrichment, introduction of toxicants) brought about by human activities. While many biological approaches have been used to measure the relationships between water quality and biological changes, the biotic index holds great promise in providing a rapid, versatile and reliable pollution index for use in a Kansas assessment program.

As a result of our evaluation process the biotic index formulation first used by Chutter (1972) was recognized as having a solid ecological basis, proven reliability, adaptability, and practical utility. These characteristics were thought to be highly desirable attributes for a proposed biotic index for use in the varied stream types and conditions present in Kansas. While we have recommended the use of the Chutter index formulation, we also recognize that this index may have to be modified for use in Kansas streams. Our biotic index review, the perceived distribution of tolerance values among taxa, and a preliminary examination of performance of the basic index in several small Kansas streams, suggest that formulation modifications may be necessary to better differentiate between water quality conditions.

State regulatory agencies and several empirical studies where the Chutter-Hilsenhoff biotic index has been used have indicated a "scale of impact" for resulting biotic index values (Table 10). Although the range of possible BI values are divided into different numbers of impact levels by various workers, there is general agreement that BI values <1.75-2.0 will be from unimpacted waters and BI values >3.75 will be from impacted sites.

Jones *et al*. (1981) evaluated water quality in Missouri Ozark streams and found high correlations between water chemistry data and relative BI values and support for their *a priori* opinions of water quality for 10 sites (eight streams). Based on statistical differences found among the sites, they fit the data into four impact levels thereby modifying Hilsenhoff's (1977) guidelines for five categories into four. Rabeni *et al*. (1985) empirically derived tolerance values from a study of 11 stream sites in Maine variously polluted with paper pulp and municipal effluents including some reference sites upstream and downstream. They used multivariate statistical methods to group sites into four groups based on community similarities. BI values also fell into four discrete groups although they did not interpret the relative degree of impact between the two middle groups.

Vermont has incorporated the Chutter-Hilsenhoff BI as part of their compliance procedures for monitoring aquatic biota since legislative amendments were made in 1986 (Act 199, Senate Bill S-42)(pers. communication, Douglas Burnham, Vt. Dept. of Water Resources, Waterbury, Vt.). The Vermont protocol gives five degrees of water quality indicated by BI values. It is not clear whether their proposed scheme for interpretation of impacts has been tested yet. Their compliance protocol, however, states that a change of 0.5 BI units indicates a possible impact has occurred. The protocol also relies on other parameters such as community similarity, EPT values (Ephemeroptera-Plecoptera-Trichoptera relative abundance), and changes in total abundance. The methodology and interpretations are aimed at detecting changes at impacted sites by comparisons with appropriate control sites made at the same time. It should be noted that this protocol calls for use of five replicate artificial substrate samplers placed in riffle areas and allowed to colonize for 6-8 weeks. Vermont also used biotic index calculations when semi-quantitative macroinvertebrate sampling is done on many streams throughout the state on a routine basis as part of their Ambient Bio-monitoring Network (ABN) program. The focus of the ABN is to determine if major qualitative changes have occurred over longer (*i.e.*, years) periods of time. The ABN is set up to help aid in determining effects from future development or impacts. Several of the guidelines for gathering baseline macroinvertebrate data include: samples are taken annually and only in the fall; only riffles are sampled; 2-6 Surber net, D-frame net and/or dredge hauls will be combined to form one sample; Surber samples are preferred; samples are in duplicate from each site and should have 400-500 macroinvertebrates. Besides calculation of a BI, taxon richness, Shannon diversity, microhabitat and feeding types of the macroinvertebrates are recorded.

New York and Massachusetts use an identical scale of four discrete levels of impact which they have found corresponds well with three other biological parameters: species richness; EPT values; and dominant species information (*i.e.*, abundance of the five most dominant species along with assessment of their known tolerance and feeding habits). This set of biological measures is part of the "rapid biological assessment" techniques developed by New York and used by both states for the past 3–4yrs (pers. communications, Arthur Johnson, Office of Environmental Affairs, Dept. of Environ. Quality Eng., Division of Water Pollution Control, Westview Building, Lyman School, Westborough, Ma.; and Robert Bode, Stream Biomonitoring Unit, U.S.P.O. Box 1397, Albany, New York).

The Illinois EPA defined five levels of impact (IEPA 1986). However, they state that only three are predicted accurately with the macroinvertebrate biotic index and suggest that other biological indicator species (*e.g.*, fish) are necessary to distinguish among low impact areas.

Hilsenhoff (1987) continuing his work in Wisconsin streams reassigned tolerance values from a 0–5 to 0–10 scale and presented an impact scale of BI values discriminating seven levels of water quality and degrees of organic pollutant impacts. It may be that the additional information since 1979 and more stream sites (>1000) has enabled an accurate distinction of seven categories of relative water quality. It is probably premature to assume BI values outside of the Wisconsin streams that Hilsenhoff has been studying would similarly fall into these seven categories.

We suggest that what is needed for Kansas is a large regional database relating BI values to known water quality assessments and empirically derived estimates of inherent variation in BI values. Certainly, BI values from a six point tolerance scale should not be divided into ≥ 6 levels of impact. It is probably best that the number of interpretable impact levels be made less than the number of tolerance values that could be confidently assigned. The number of impact levels could be based on the number of tolerance categories where distinctions between adjacent tolerance values (or groups of several tolerance values) were clearly correlated to an interpretable degree of pollutant impact. There are a variety of specific approaches (although we will not discuss these further here) that might be taken to determine an appropriate scale of interpretation for BI values from Kansas streams and rivers. In addition we believe it is most important that assessments of relative tolerance should be based on regionally derived empirical data. The high number of taxa assigned intermediate tolerance values (especially 2 and 3) may cause biotic index values to compress near the central portion of the 0-5 integer scale. This potential phenomenon may affect interpretations, and result in the failure to utilize the broader range of possible scores (values). Our initial modifications with a weighing factor for sensitive taxa which was incorporated into the basic formula causes impacted and unimpacted site scores to diverge without the loss of the group effect offered by the original index. The use of sensitive taxa in this manner is compatible with the well documented responses of organisms sensitive to organic pollution. We are encouraged in our investigation of this approach by the finding of researchers from Denmark. The use of positive index (sensitive) and negative index (tolerant) groups in their index scheme for identification of organic pollution allows better separation of mid-range values.

Tolerance values for those organisms in the mid-tolerance range (2-3) should be reviewed and evaluated as to their value as "indicator" taxa. Assignment of tolerance values 2 and 3 to specific taxa was sometimes subjective because of the lack of data from which more critical judgments could be obtained. It has become clear that within the taxa receiving tolerance values of 2 or 3 there exists two somewhat distinct types of organisms and responses:

1) Species which tend to tolerate conditions through a broad range of pollution conditions. For example, Species A may be associated with unimpacted waters but it may also survive under moderately impacted stream conditions and a value of 2 or 3 might seem appropriate to indicate its tolerance "limit". Species A could be termed "facultative" and a tolerance value of 2 or 3 only reflects the upper limit of water quality conditions in which it will occur.
2) Species which may actually benefit from conditions associated with moderately polluted water. Suppose Species B is found predominantly in a narrow range of

> intermediate water quality conditions. A tolerance value of 2 or 3 for Species B would then indicate its preference for or selective tolerance only to “moderate” stream conditions since Species B would not be expected to occur in stream conditions either less or more polluted.

The identification and separation of those organisms that display more “facultative” responses like Species A from those that are more restrictive “indicator” species like Species B may enhance the performance of the biotic index. It is within these intermediate values that most Kansas taxa were placed and their potential impact on the biotic index is obvious.

The use of an expanded scale (*e.g.*, 0-10) may increase the sensitivity of the Chutter index. Both Chutter (1972) and eventually Hilsenhoff (1987) used a 0-10 tolerance value scheme. If we examine the development and refinement process that Hilsenhoff followed, we are quick to realize that the process of evaluating or re-evaluating tolerance values for a 0-10 value scale is costly. Only after 10 years of work and the examination of >1000 stream samples was Hilsenhoff able to propose a change from his original 0-5 scale (Hilsenhoff 1977) to his current 0-10 scale (Hilsenhoff 1987). A concerted effort would have to be made in Kansas if our currently proposed 0-5 scale was to be expanded. It is our belief that the most logical scale expansion might be within the NOD category and would come from rather intense studies on stream reaches known to exhibit high nutrient loadings but relatively free from other major pollutants (*e.g.*, heavy metals, pesticides).

Another area of concern in regard to the basic Chutter formula is the use of abundance data obtained from the samples. Typically invertebrate communities are dominated by a few highly abundant taxa. The advantages or effects of enumerating those taxa that can occur in extreme numbers (often a magnitude greater than most other taxa) on a calculated biotic index should be investigated. The occurrence of any one species in great numbers could mitigate the importance of other indicator species. The added cost in manpower and time required in total enumeration of those few highly abundant taxa should be evaluated against the informational loss or gain resulting from total counts and, perhaps, an "upper limit" abundance category could be defined.

Verification of many of the tolerance values with empirical data from field studies is considered a necessity. Field studies should be designed such that potential variables (*e.g.*, temporal, habitat) may be accounted for when examining the effects of pollution on various taxa. The use of existing field information and studies may prove to be of some value but often these types of data lack the continuity or experimental design necessary to provide meaningful tolerance values.

Quantification of the relationship between the Habitat Development Index (HDI) and Biotic Index (BI) values in both unimpacted and impacted streams must be established. If a relationship exists and can be quantified, it may be possible to adjust the biotic index for habitat differences. Ideally, independence from habitat influences are desirable in a biotic index but our review indicates that such is not the case and most indices restrict themselves to specific habitat sampling. This restricted or selective habitat sampling approach cannot be used in Kansas because of the extremely diverse nature of stream types found in this state. If HDI and BI relationships cannot be established, the use of artificial substrates may provide a proven alternate approach (*e.g.*, Hawkes 1977; DePauw *et al*. 1986).

The within site variability of biotic index values associated with a particular assessment site has not been addressed in the body of this paper. However, it is a reality and must be considered by potential users. Within a site the variance in the BI depends on the spatial distribution of aquatic insects and the affects of temporal fluctuations on the composition of the insect community. Jones *et al*. (1981) suggests that in Ozark streams sampled by "kick-net" sampling in riffles about five samples were necessary to identify spatial biases so that statistically significant differences in BI values between sites could be obtained. Other work done to verify the statistical reliability of the Hilsenhoff index and the reproducibility of the sample collections and sorting procedures can be found in Eilers (1980), Hilsenhoff (1982) and Narf *et al*. (1984).

Temporal variations in biotic index values have been recognized in almost all proposed indices including the work of Hilsenhoff (1982, 1987), Chutter (1972), Murphy (1978), Chester, (1980), Jones *et al*. (1981) and DePauw *et al* (1986). Hilsenhoff's work has suggested that a temporal correction factor can be obtained so the BI's taken during different seasons can be standardized (Hilsenhoff 1977). However, both Murphy (1978) and Jones *et al*. (1981) both found greater temporal variability when assessing river water quality with indices based on community diversity than with biotic indices. Temporal effects on a Kansas biotic index will need to be identified.

Many workers and States (*e.g.*, Vermont) suggest that for monitoring and assessment purposes, macroinvertebrate samples be collected during the fall period (Sept-Oct). The rationale is: (1) communities at this time would still reflect conditions of the late summer stress period; (2) few species are hatching at this time, thus the communities are more stable, allowing better inter- and intra-site comparisons; and (3) most larval forms are further developed in the fall than during the midsummer period facilitating better taxonomic resolution. We have some reservations concerning fall sampling in Kansas. In Kansas many organisms have evolved somewhat complex life cycles to handle the naturally occurring harsh low-flow conditions that are common to many of our streams. Species may be present in forms that cannot be sampled because of delayed development (*e.g.*, diapausing eggs or larvae). A spring sampling program may be better suited for assessment purposes in Kansas.

Our investigation of the literature revealed that most workers agree that the preferred sampling periods are spring and autumn although other seasons may be considered (*e.g.*, DePauw et. al. 1986; Armitage *et al*. 1983). The comprehensive investigations by Murphy (1978) and Armitage *et al*. (1983) clearly showed that spring values were consistently higher than other season values and that temporal variations can mask spatial differences.

We offer a closing remark concerning the use of a biotic index to assess water quality conditions in Kansas. The proposed biotic index scheme should prove extremely useful in providing a rapid, cost-effective method of biological assessment. Its use within a comprehensive bioassessment program is highly recommended.

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TABLES

Table 1. Beak's river index (modified from Beak 1965).

Pollution status	**Biotic index**	**Type of macroinvertebrate community**	**Fisheries potential**
Unpolluted	6	Sensitive, facultative and tolerant predators, herbivores, filter and detritus feeders all represented. No species well developed.	All normal fisheries for type of water
Slight to moderate pollution	5 or 4	Sensitive predators and herbivores reduced in population density or absent. Facultative predators, and possibly filter and detritus feeders well developed and increasing in numbers as index decreases	Most sensitive fish species reduced in numbers or missing
Moderate pollution	3	All sensitive species absent and facultative predators (Hirudinea) absent or scarce. Predators of family Tanypodinae and herbivores Chironomidae present in fairly large population densities.	Only coarse fisheries maintained
Moderate to heavy pollution	2	Facultative and tolerant species in numbers if pollution toxic; if organic, a few species insensitive to low oxygen present in large numbers	If fish present, only those with high tolerance of pollution
Heavy pollution	1	Only most tolerant detritus feeders (Tubificidae) present in large numbers	Very little, if any, fisheries
Severe pollution usually toxic	0	No macroinvertebrates present	No fish

Table 2. Biological Monitoring Working Party (BMWP) Score (from Hellawell 1986).

Families	**Score**
Siphlonuridae, Heptageniidae, Leptophlebiidae, Ephemerellidae, Potamanthidae, Ephemeridae Taeniopterygidae, Leuctridae, Capniidae, Perlodidae, Perlidae, Chloroperlidae Aphelocheiridae Phryganeidae, Molannidae, Beraeidae, Odontoceridae, Leptoceridae, Goeridae, Lepidostomatidae, Brachycentridae, Sericostomatidae	10
Astacidae Lestidae, Agriidae, Gomphidae, Cordulegasteridae, Aeshnidae, Cordulliidae, Libellulidae Psychomyiidae, Philopotamidae	8
Caenidae Nemouridae Rhyacophilidae, Polycentropodidae, Limnephilidae	7
Neritidae, Viviparidae, Ancylidae Hydroptilidae Unionidae Corophiidae, Gammaridae Platycnemididae, Coenagriidae	6
Mesovelidae, Hydrometridae, Gerridae, Nepidae, Naucoridae, Notonectidae, Pleidae, Corixidae Haliplidae, Hygrobiidae, Dytiscidae, Gyrinidae, Hydrophilidae, Clambidae, Helodidae, Dryopidae, Eliminthidae, Chrysomelidae, Curculionidae Hydropsychidae Tipulidae, Simuliidae Planariidae, Dendrocoelidae	5
Baetidae Sialidae Piscicolidae	4
Valvatidae, Hydrobiidae, Lymnaeidae, Physidae, Planorbidae, Sphaeriidae Glossiphoniidae, Hirudidae, Erpobdellidae Asellidae	3
Chironomidae	2
Oligochaeta (whole class)	1

Table 3. "Indication" groups and weighted scores from the Chandler score system as proposed by Chandler (1970) (refer to text for abundance levels).

		Increasing Abundance Weighted Scores				
	Groups present in sample	P	F	C	A	V
Each species of:	*Planaria alpina*, Taenopterygidae, Perlodidae, Isoperlidae, Perlidae, Chloroperlidae	90	94	98	99	100
Each species of:	Leuctridae, Capniidae, Nemouridae (exclud. *Amphinemura*)	84	89	94	97	98
Each species of:	Ephemeroptera (exclud. *Baetis*)	79	84	90	94	97
	Cased Trichoptera, Megaloptera	75	80	86	91	94
	Ancylus	70	75	82	87	91
	Rhyacophila (Trichoptera)	65	70	77	83	88
Genera of:	*Dicranota*, *Limnophera*	60	65	72	78	84
	Simulium	56	61	67	73	75
	Coleoptera, Nematoda	51	55	61	66	72
	Amphinemura (Plecoptera)	47	50	54	58	63
	Baetis (Ephemeroptera)	44	46	48	50	52
	Gammarus	*40*	*40*	*40*	*40*	*40*
	Uncased Trichoptera (exlud. *Rhyacophila*)	38	36	35	33	31
	Tricladida (exclud. *P. alpina*)	35	33	31	29	25
Genera of:	Hydracarina	32	30	28	25	21
Each species of:	Mollusca (exclud. *Ancylus*)	30	28	25	22	18
Each species of:	Chironomidae (excluding *C. riparius*)	28	25	21	18	15
	Glossiphonia	26	23	20	16	13
Each species of:	*Asellus*	25	22	18	14	10
	Leech (exclud. *Haemopsis*, *Glossiphonia*)	24	20	16	12	8
	Haemopsis	*24*	*20*	*16*	*10*	*7*
	Tubifex	*22*	*18*	*13*	*12*	*9*
	Chironomus riparius	21	17	12	7	4
	Nais	*20*	*16*	*10*	*6*	*2*
Each species of:	Air breathing species	19	15	9	5	1
	No animal life	0	0	0	0	0

Table 4. Chutter's interpretation of the cleanliness of South African rivers based on his biotic index values (Chutter 1972).

Biotic Index Value	**Interpretation**
0-2	Clean, unpolluted waters
2-4	Slightly enriched waters, the slight enrichment may be due either to the natural occurrence of organic matter or to high quality effluents containing a little organic matter or its breakdown products. Chemical changes in the water may be hardly detectable.
4-7	Enriched waters, the higher a biotic index value, the greater the enrichment. Obvious increases in BOD and nitrogenous compounds in the water, and rather wide diurnal fluctuations in dissolved oxygen are to be expected.
7-10	Polluted waters in which there will be great increases in chemical parameters associated with organic pollution.

Table 5. Classification of streams by average of 1977 and 1978 biotic index values (Hilsenhoff 1982).

Biotic Index	**Water Quality***	**# of streams in category**
≤ 1.75	Excellent	18
1.75 - 2.25	Very good	12
2.26 - 2.75	Good	14
2.76 - 3.50	Fair	6
3.51 - 4.25	Poor	1
≥ 4.26	Very poor	1

* Water quality apparently refers to organic enrichment or disturbance

Table 6. Practical limits to determine systematic units to be used in the Belgian biotic index (from DePauw and Vanhooren 1983).

Taxonomic Groups	**Systematic Units**
Non-insecta	
Plathelminthes	genus
Oligochaeta	family
Hirudinea	genus
Mollusca	genus
Crustacea	family
Hydracarina	presence
Insecta	
Plecoptera, Ephemeroptera, Odonata, Megaloptera & Hemiptera	genus
Trichoptera, Coleoptera	family
Diptera (except Chironomidae)	family
Diptera: Chironomidae	*thumni-plumosus* group Non-*thumni-plumosus* group

Table 7. Standard table to determine the Belgian biotic index (modified from DePauw and Vanhooren 1983).

Column I (Faunistic groups)	**Column II (Number of SU/group)***	**Column III (Total numbers of systematic units present in sample)**				
		0-1	2-5	6-10	11-15	16 and more
		Biotic index values				
1. Plecoptera or Ecdyonuridae (=Heptageniidae)	SU $\geq$ 2		7	8	9	10
	SU = 1	5	6	7	8	9
2. Cased Trichoptera	SU $\geq$ 2 and above SU are absent		6	7	8	9
	SU = 1 and above SU are absent	5	5	6	7	8
3. Ancylidae or Ephemeroptera except Ecdyonuridae	SU $\geq$ 3 and above SU are absent		5	6	7	8
	SU $\leq$ 2 and above SU are absent	3	4	5	6	7
4. Aphelocheirus or Odonata or Gammaridae or Mollusca (except Sphaeridae)	SU present and above SU are absent	3	4	5	6	7
5. Asellus or Hirudinea Sphaeridae or Hemiptera (except Aphelocheirus)	SU present and above SU are absent	2	3	4	5	
6. Tubificidae or Chironomidae of the *thummi-plumosus* group	SU present and above SU are absent	1	2	3		
7. Eristalinae (= Syrphidae)	SU present and above SU are absent	0	1	1		

SU = systematic units
note = If no systematic units are present in the sample, the biotic index is 0

Table 8. Biocoenotic responses of indicator value induced by pollutants (modified from Hawkes 1977).

Response	**Response Description**
A	Appearance or disappearance of individual taxa of indicator value
B	Reduction in total number of taxa of a community
C	Changes in the abundance of individual taxa
D	Changes in the relative abundance within a community
E	Changes in the degree of heterotrophy-autotrophy
F	Changes in the degree of productivity of a community

Table 9. Sample scoring form for the proposed Habitat Development Index (HDI).

Habitat Development Index

Stream________________________________ Sample No.____________________ Date________________________

County________________________________ Legal Description____

______________________________________ Evaluator____

Score only those macro and microhabitat categories that were sampled							Riffle	Pool	Run
MINIMUM MACROHABITAT SCORE	Absent: 0		Present: 3						
AVERAGE DEPTHS	Riffles	<5 cm: 0	5-10 cm: 1	>10 cm: 2					
	Pools	<30 cm: 0	30-60 cm: 1	>60 cm: 2					
	Runs	<15 cm: 0	15-45 cm: 1	>45 cm: 2					
RIFFLE SUBSTRATE SCORE	% Cobble*	0-10%: 0	11-25%: 1	26-50%: 2	>50%: 3	A=___			
	% Embeddedness	0-25%: 0	26-75%: -1	>75%: -2		B=___			
	Record score in right hand column only if A+B $\geq$ 0					A+B=____			
ORGANIC DETRITUS AND DEBRIS	No organic detritus or debris was sampled: 0	Only sparsely scattered bits of detritus were sampled: 1	Large leaf packs or large amounts of scattered detritus were sampled: 2	Both detritus and debris including logs were sampled: 3					
ALGAL MASSES	No algal masses were sampled: 0		Algal masses were sampled: 1						
MACROPHYTES	No macrophytes were sampled: 0	Very few macrophytes or small patches of plants were sampled: 1	Many macrophytes or large areas of dense growth were sampled: 2						
BANK VEGETATION	No bank vegetation was sampled: 0	Only small amounts of thin bank vegetation was sampled: 1	Submerged tree roots or thick bank vegetation was sampled: 2						
	MACROHABITAT SCORES								
	SAMPLE SCORE								

* If % cobble <=10% and boulders or bedrock present, score box A as 1.

Table 10. Levels of pollutant impact or water quality indicated by use of a macroinvertebrate biotic index following the Chutter-Hilsenhoff (1982) formulation as used by workers in various regions of the United States. (See text for citation of the sources.)

State Biotic Index Range (0-5)[1] and Levels of Impact

Missouri	0	1.75	2.5	3.25	5
	Unpolluted	Slightly enriched	Enriched	Polluted	

Maine	0	1.7	2.9	4.6	5
	Group I unaffected	Group II somewhat impacted	Group III somewhat impacted	Group IV Inhospitable	

Missouri	0	1.75	2.5	3.25	5
	Unpolluted	Slightly enriched	Enriched	Polluted	

New York and Massachusetts	0	2.0	3.0	4.0	5
	Nonimpacted	Slightly impacted	Moderately impacted	Severely impacted	

Illinois	0			3.4	4.5	5
		(cannot be determined)				
	Unique aquatic resource					
		Highly valued aquatic resource				
			Moderate aquatic resource			
				Limited aquatic resource		
					Restricted Aquatic Resource	

Vermont	0	2.0	2.5	3.0	3.5	5
	Excellent	Good	Fair	Poor	Very poor	

Wisconsin	0	1.75	2.25	2.75	3.25	3.75	4.25	5
	Excellent							
		Very good						
			Fair					
				Fairly poor				
					Poor			
						Very poor		
	No apparent organic pollution							
		Possible slight OP						
			Some OP					
				Fairly significant OP				
					Significant OP			
						Very signif OP		
							Severe OP	

[1]conversions made from original BI scales: 0-11 Illinois, 0–10 Wisconsin, and 0–3 Maine.

FIGURES

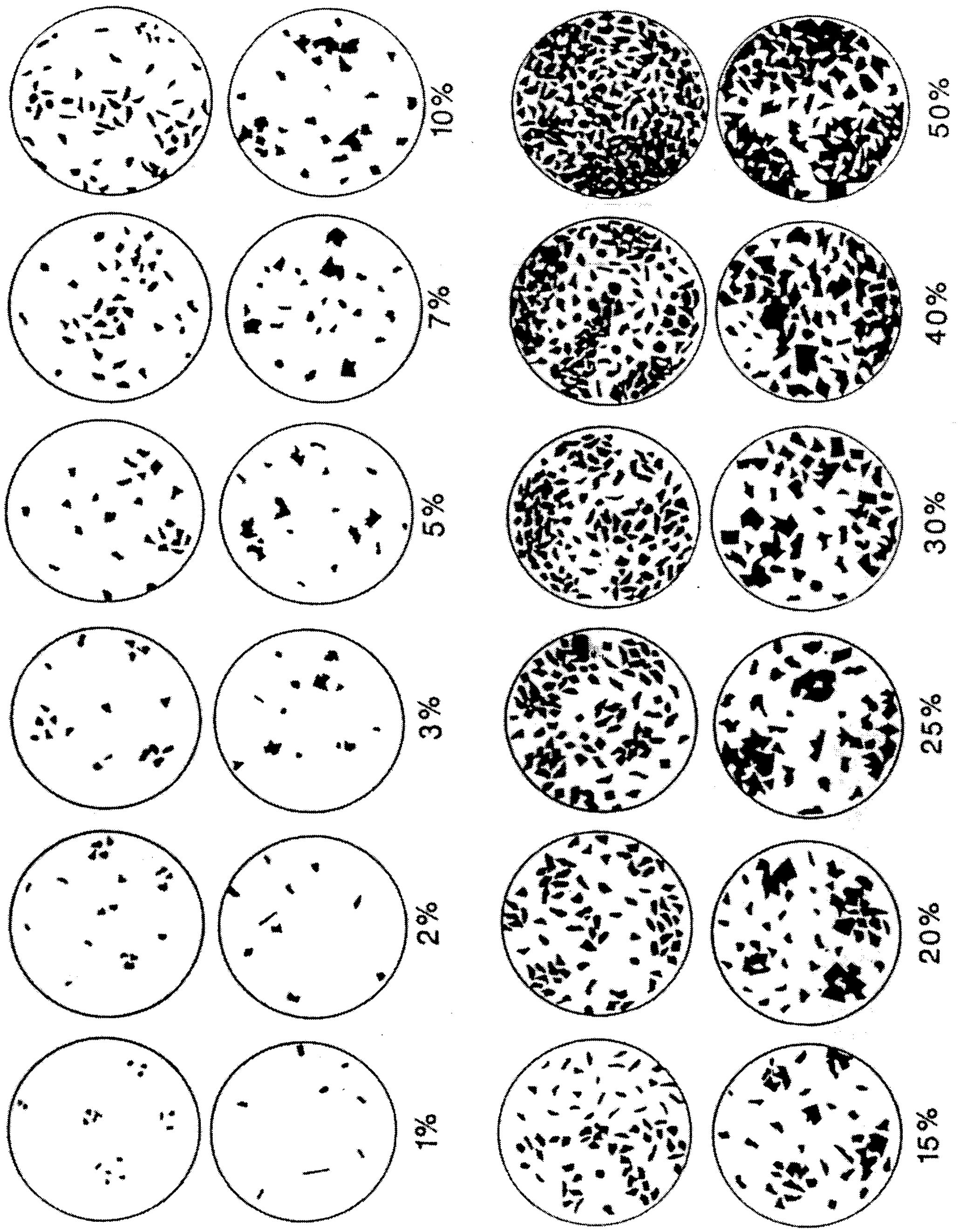
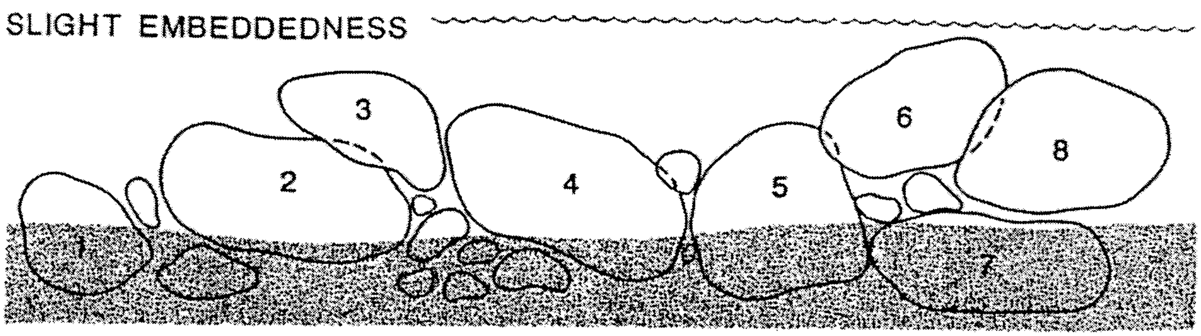
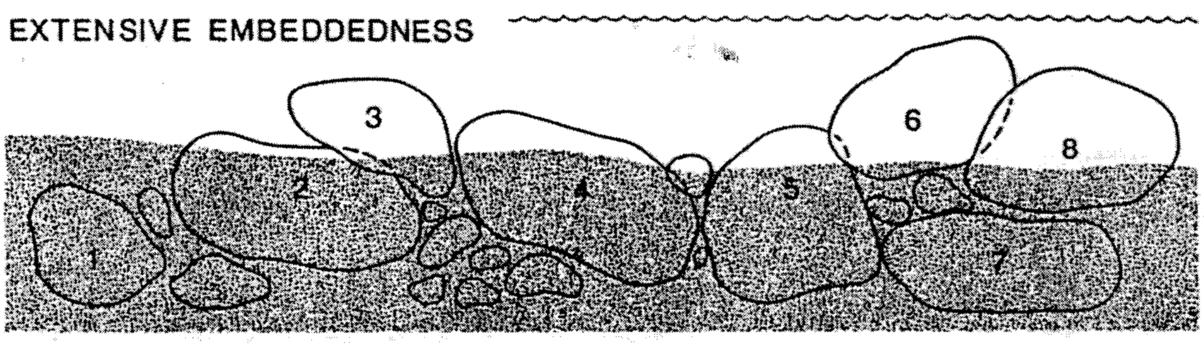


Figure 1. Estimating percent substrate that is cobble-sized (ca. 6-26cm). Darkened areas represent coverage by cobble. For HDI scoring purposes, estimates need only be one of four choices 0-10%, 11-25%, 26-50%, or > 50%.



Individual Cobble	% Embedded (approx)	Score
1	60	-1
2	15	0
3	0	0
4	20	0
5	40	-1
6	0	0

HDI Embeddedness scor e= 0



Individual Cobble	% Embedded (approx)	Score
2	90	-2
3	20	0
4	80	-2
5	10	-2
6	30	0
8		-1

HDI Embeddedness score = -1

Figure 2. Examples of two conditions of embeddedness (slight and extensive). Cobble-sized stones are numbered. An HDI embeddedness score is based upon the predominant condition (score) of embeddedness given after examination of six or more surface-occurring cobble-sized stones. Individual embeddedness scores are 0, -1, or -2 for 0-25%, 26-75%, or > 75% embeddedness, respectively.

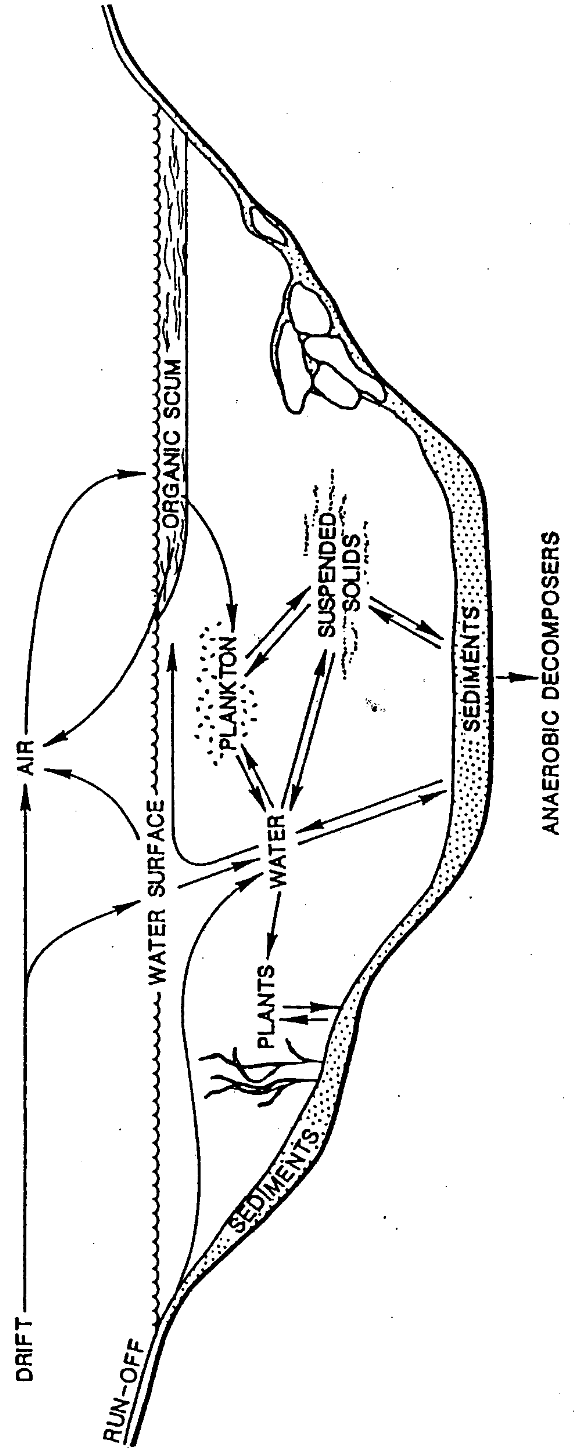


Figure 3. Patterns of pesticide partitioning in streams (modified from Edwards (1977)).

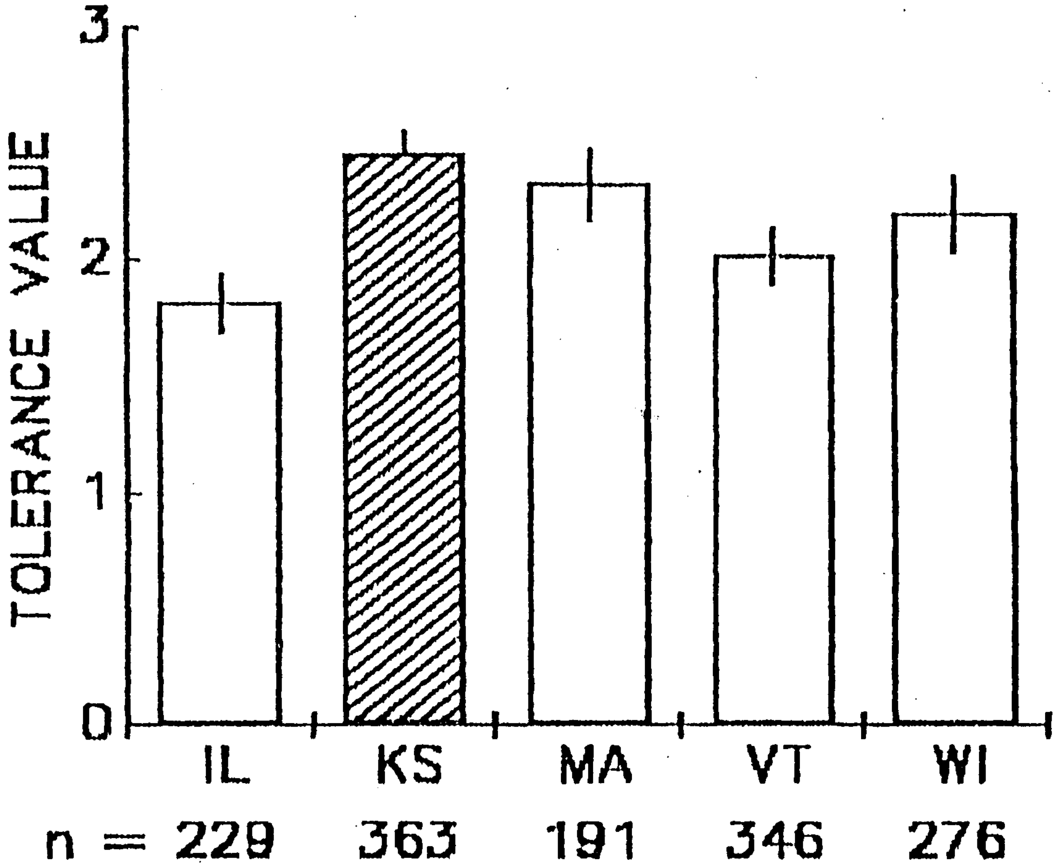


Figure 4. Overall mean tolerance values (and 95% confidence intervals) for genera in six aquatic insect orders as found in each of five states. Orders included in the means were Plecoptera, Trichoptera, Ephemeroptera, Coleoptera, Odonata, and Diptera. n = the total number of genera for these six orders that were given tolerance values in each state. Shaded column is the mean of our proposed tolerance values for Kansas genera.

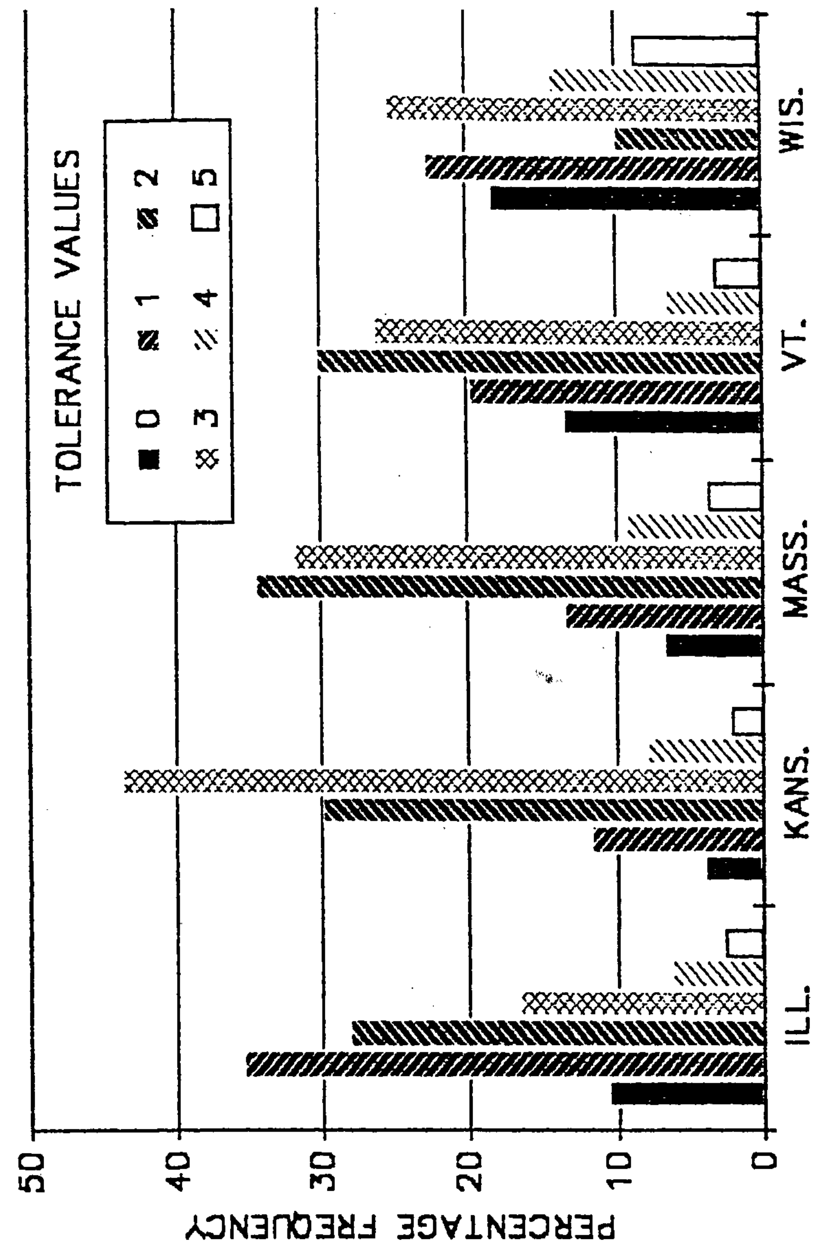


Figure 5. Frequency (percent) distributions of tolerance values on a six point integer (0-5) scale as assigned among genera in six aquatic insect orders found in each of five states. Orders included were as in Figure 4.

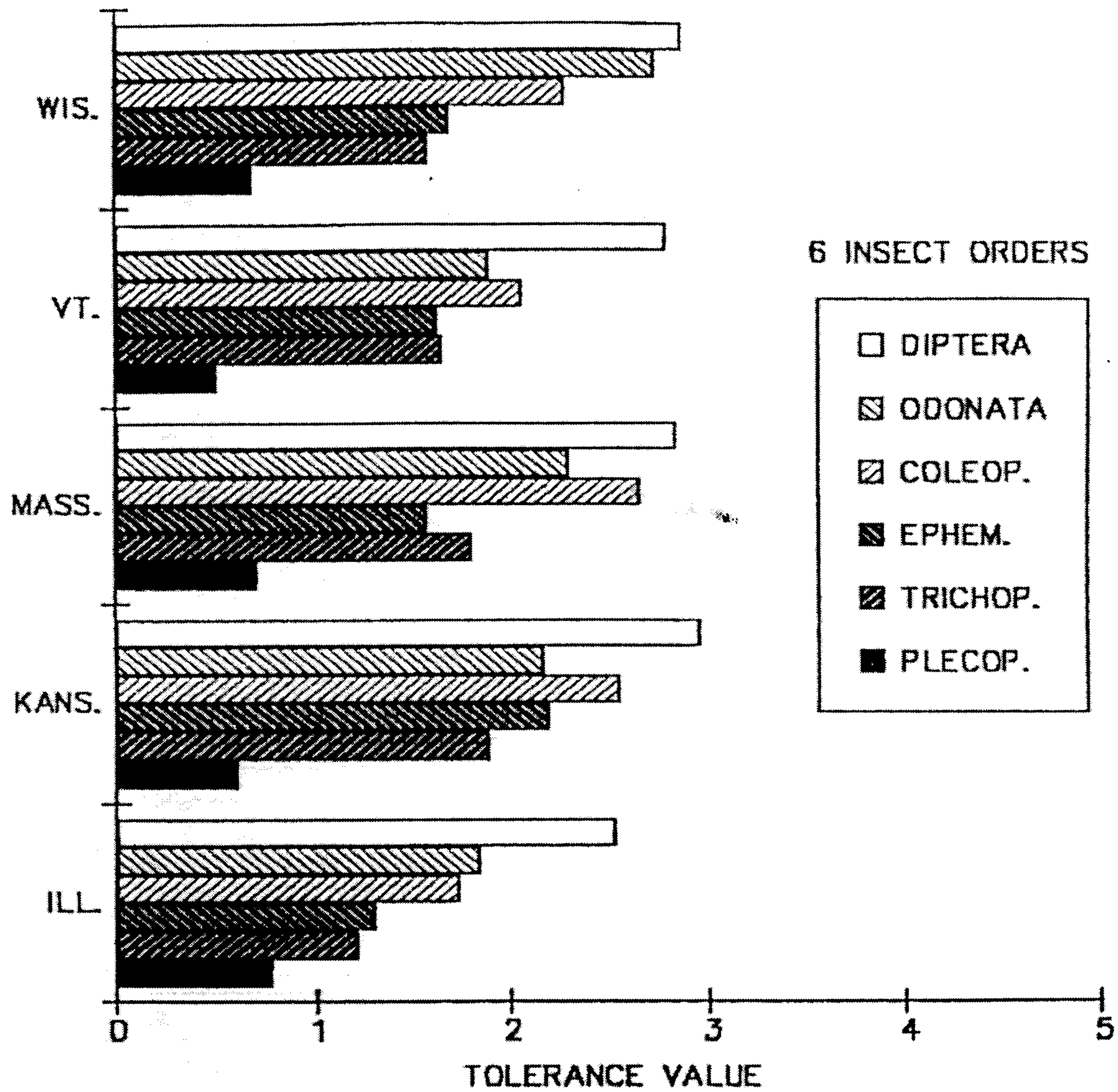


Figure 6. Mean tolerance values for genera in each of six aquatic insect orders as found in each of five states.

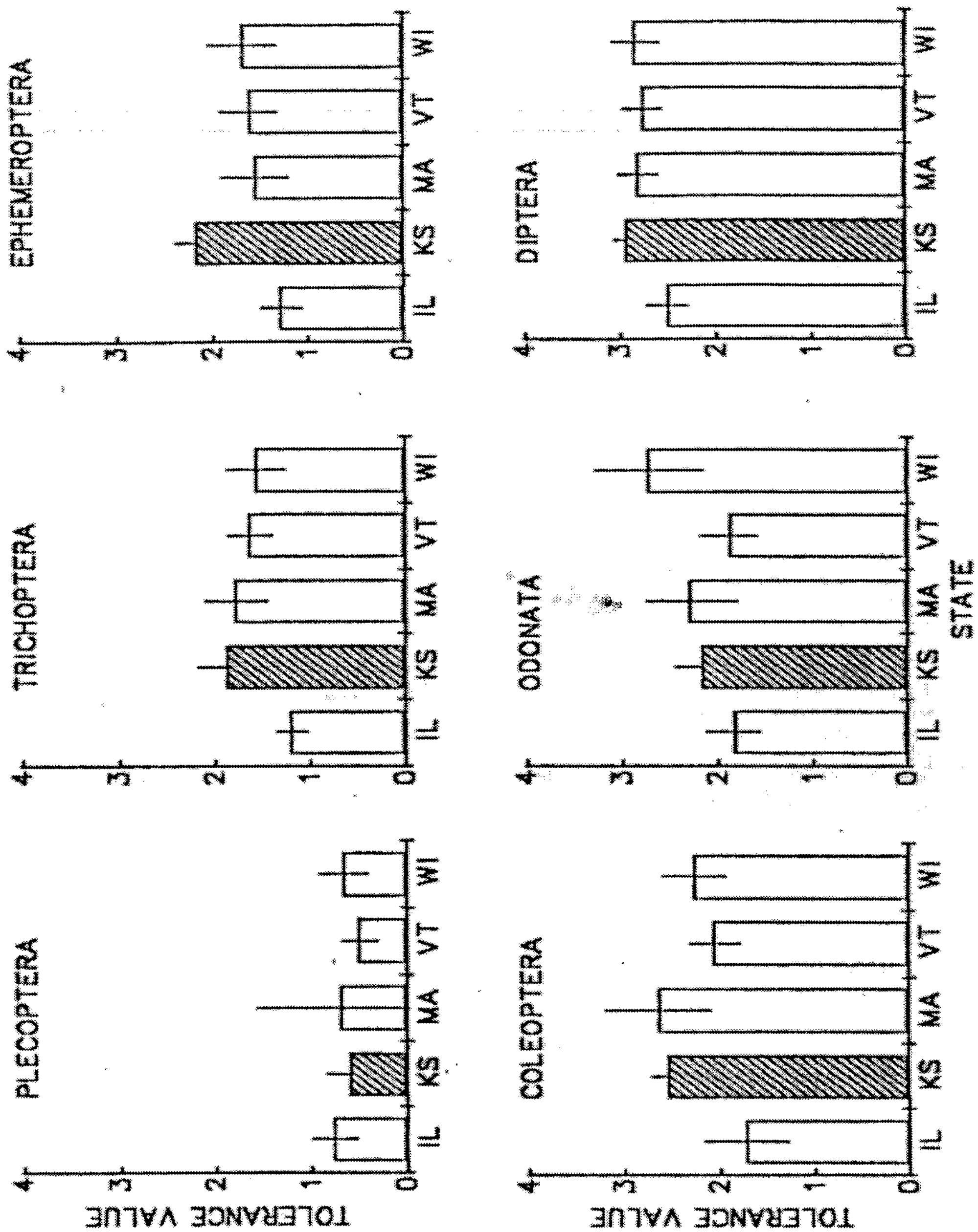


Figure 7. Mean tolerance values (and 95% confidence intervals) for genera in each of five states for six aquatic insect orders. Shaded columns are the means of our proposed tolerance values for Kansas genera.

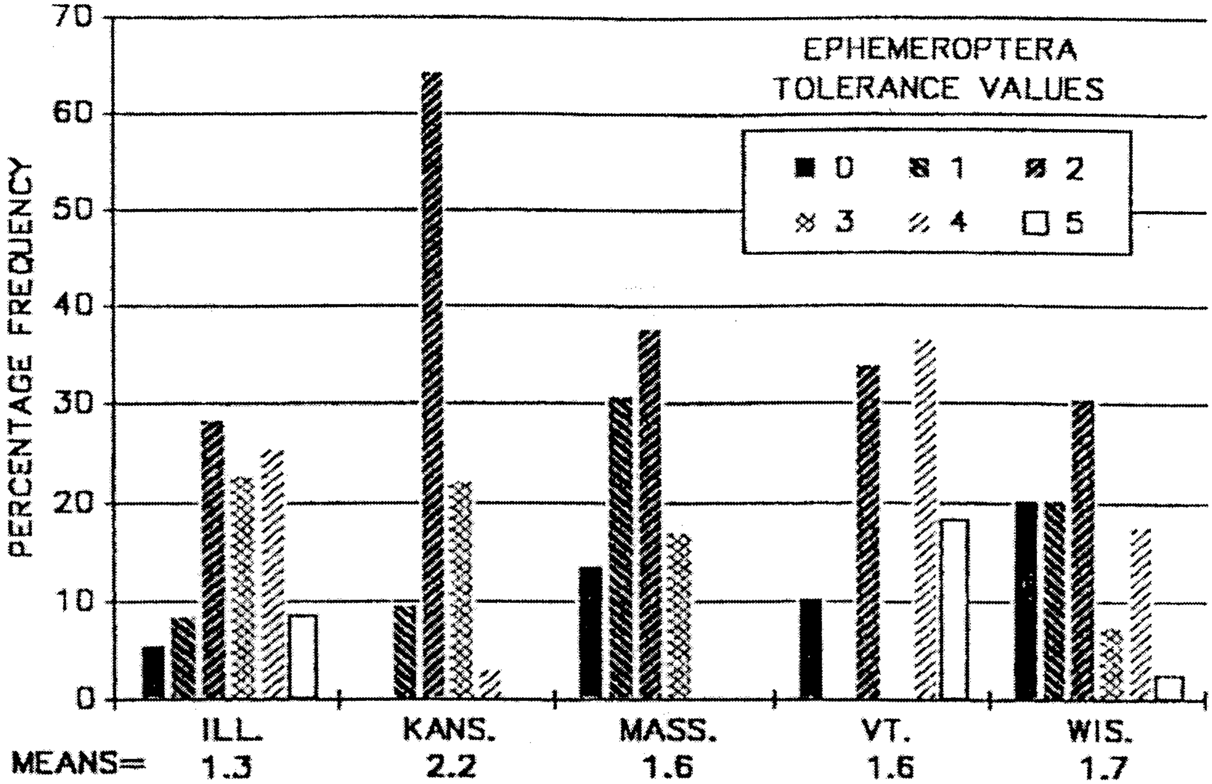


Figure 8. Frequency (percent) distributions of tolerance values on a six point integer (0-5) scale as assigned to the genera for the order Ephemeroptera found in each of five states.

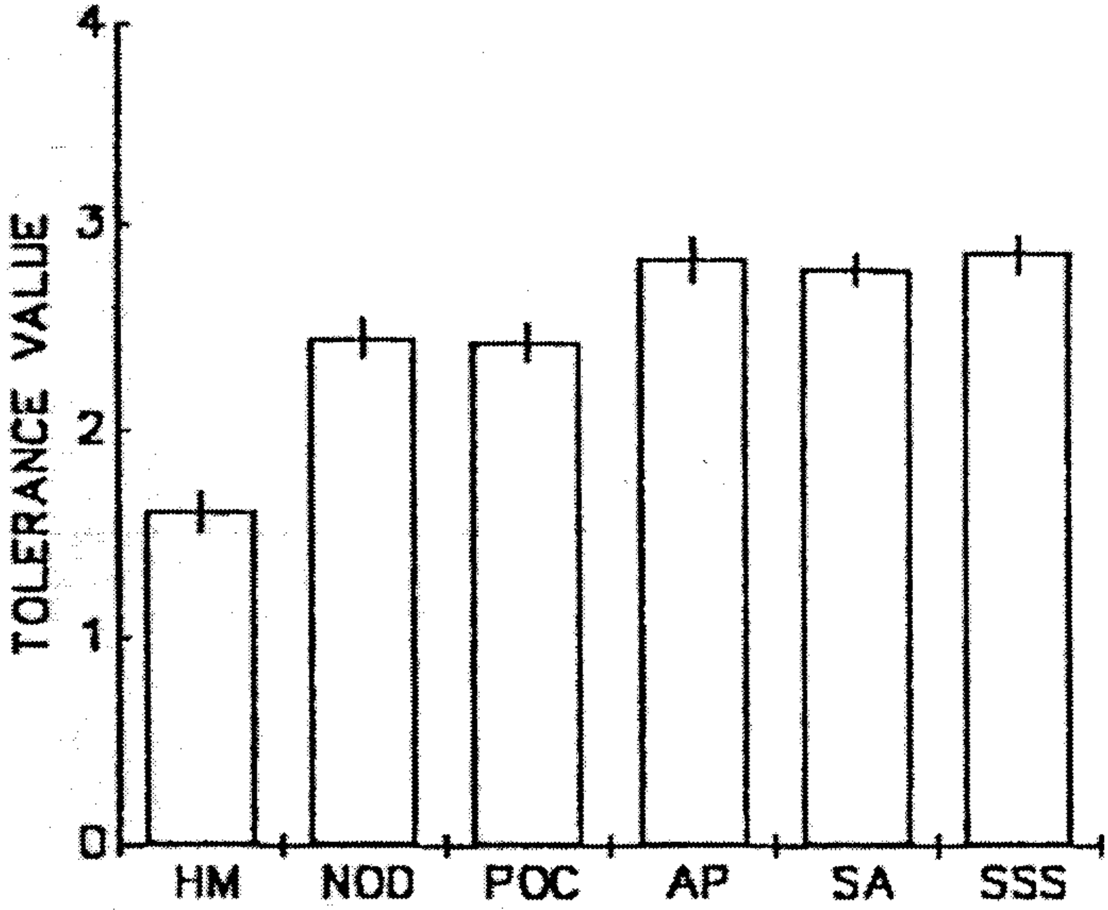


Figure 9. Overall mean tolerance values (and 95% confidence intervals) for genera in Kansas in six aquatic insect orders as assigned for each of six pollutant categories. Pollutant categories were HM = heavy metals; NOD = nutrient and oxygen-demanding substances; POC = persistent organic compounds; AP = agricultural pesticides; SA = salinity; SSS = suspended solids and sediments.

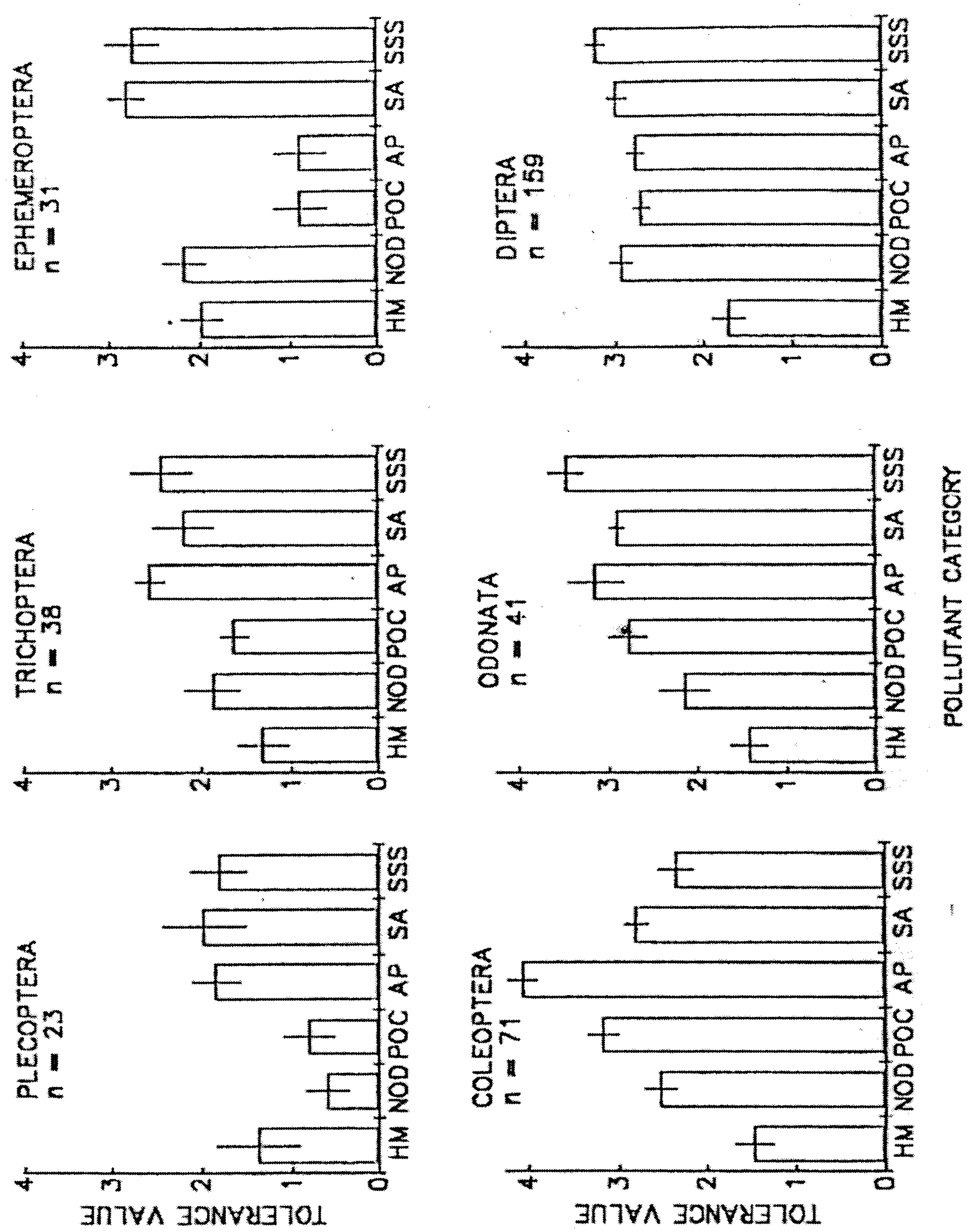


Figure 10. Mean tolerance values (and 95% confidence intervals) for genera in each of six pollutant categories for six aquatic insect orders. n = number of genera in each of the orders.

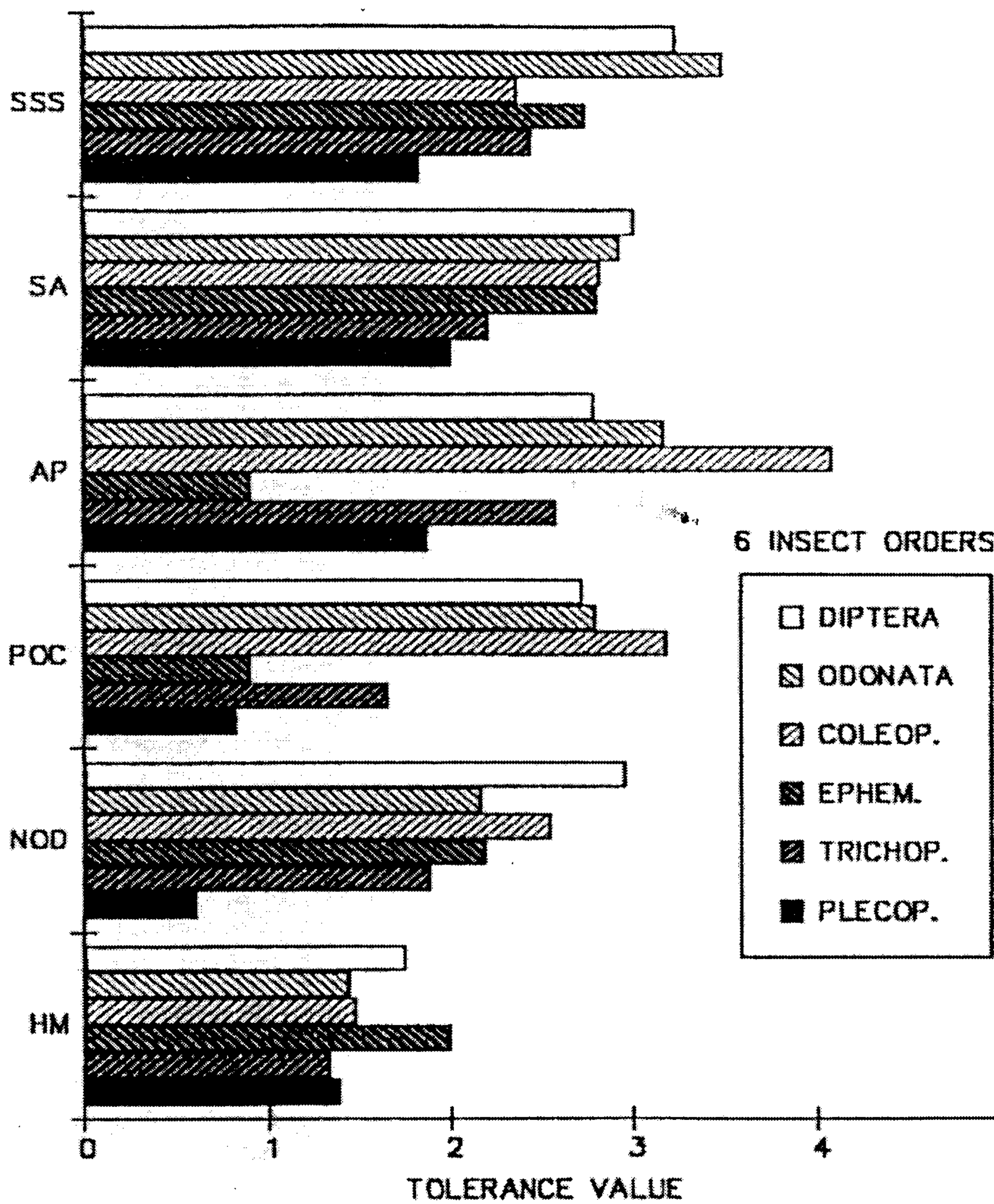


Figure 11. Mean tolerance values for genera in each of six aquatic insect orders as assigned for each of six pollutant categories. Pollutant categories were as in Figure 9.

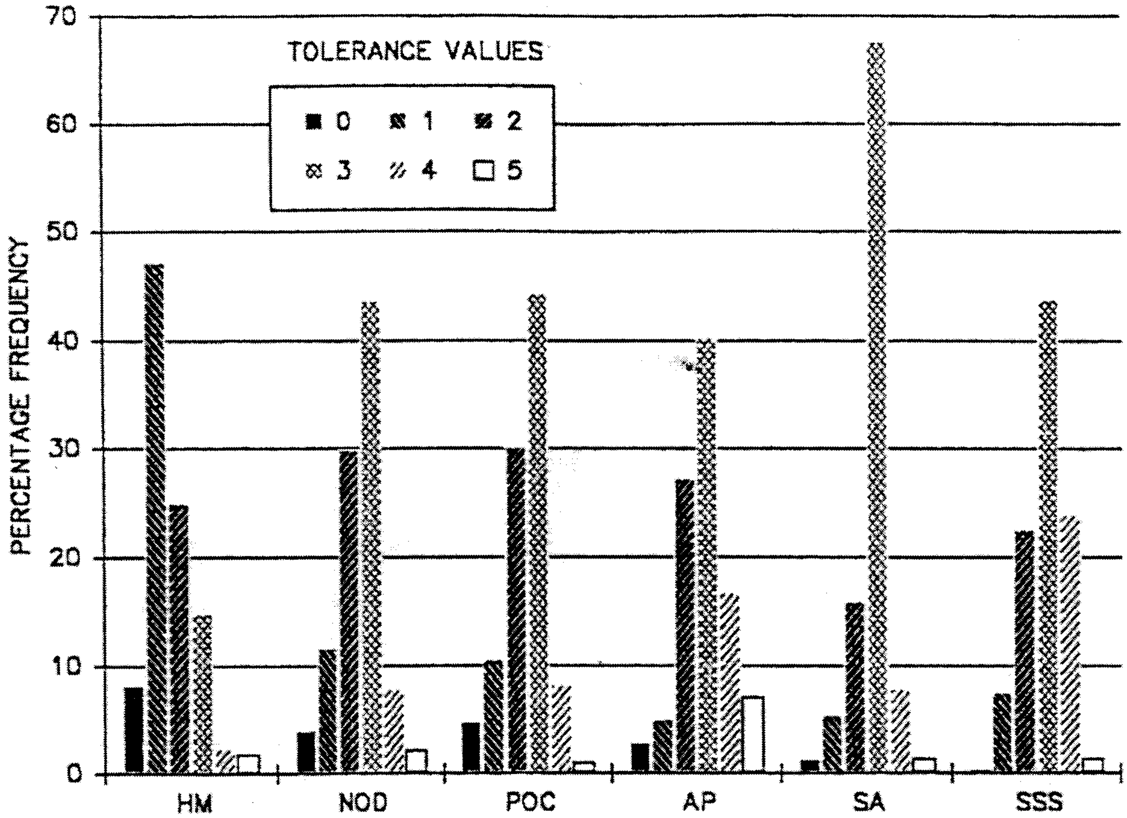


Figure 12. Frequency (percent) distribution of proposed tolerance values on a six point integer (0-5) scale as assigned among genera in six aquatic insect orders. Distributions are for each of six pollutant categories as in Figure 9. Insect orders included were as in Figure 4.

APPENDIX I. – Sample Questionnaire and Responses

A sample questionnaire and our cover letter sent to regulatory agencies in 50 states followed by cover letters and the questionnaires we received from the 28 states that responded. *[Note: Responses are NOT included in the newly reformatted version of this document.]*

KANSAS BIOLOGICAL SURVEY

The University of Kansas

Raymond Nichols Hall
2291 Irving Hill Drive—Campus West
Lawrence, Kansas 66045-2969
(913) 864-4777

February 20, 1987

Dear Sir or Madam:

The Kansas Biological Survey is currently involved in the formulation of a Biotic Index to rate the quality of Kansas streams and rivers. Ideally the final Biotic Index will be sensitive to organism response to the following types of perturbations:

No = Nutrient and Oxygen demanding substances. (NO) (e.g. Sewage effluent)
AP = Agricultural pesticides. (Atrazine, Round-up)
HM = Heavy Metals. (Lead, Zinc, Iron)
ROC = Refractory organic compounds. (PCB's, DDT, and other older pesticides such as Chlordane)
VOC = Volatile organic compounds.
DS = Dissolved solids. (Chlorides, Sulfates, Phosphates)
SSS = Suspended solids and sediments.

The checklists provided in this packet contain a listing of Kansas taxa with a biotic index value given in the first column at the right (RT). The other seven columns are listings of the above mentioned perturbance factors. Please assign your best estimate by the following rating system:

0=Intolerant	3=Intermediate
1=Intermediate	4=Intermediate
2=Intermediate	5=Very Tolerant

A value of 0 might be assigned an organism whose populations are eliminated in a stream containing low concentrations of the particular pollutant and a value of 5 might indicate the presence of good populations of a organism even when exposed to rather high levels of the above mentioned pollutants. Intermediate conditions might be used to judge tolerance if other data is lacking. If you have no idea simply enter a question (?) mark or no mark in to the appropriate space.

Also, enclosed is a questionaire about methods possibly being utilized by your state. It would be greatly appreciated if you could spend a few minutes giving us your comments and suggestions.

Thank you,

Donald G. Huggins
Associate Scientist

Questionnaire

1. Is your state currently using a Biotic Index (*e.g.* Hilsenhoffs BI) to measure water quality?
2. If so, may we obtain a copy of the system presently being implemented in your state?
3. If not, are you currently utilizing another system such as dicersity indices, similarity indices, *etc.*? What biotic evaluation process(es) are utilized?
4. How many years have you been using your current evaluation system?
5. In your opinion, how effective is the biotic evalution system presently in use in your state in terms of identifying biological disturbance?
6. Are assessment capabilities adequate?
7. What are the main problem areas associated with implementation?
8. Are biological assessments made in conjunction with chemical/physical evaluations?
9. By and large, how are tolerance values assigned if a BI is used (literature values, judgement and person experience, own research, combinations of information)?
10. Other comments.

APPENDIX II. – List of Proposed Tolerance Values

Lists of proposed tolerance values on a six point integer (0-5) scale for taxa in 10 orders of aquatic insects known to occur in Kansas. The lists presented are for the orders: Diptera, Coleoptera, Ephemeroptera, Hemiptera, Lepidoptera, Megaloptera, Neuroptera, Odonata, Plecoptera, Trichoptera. Each list provides tentative tolerance values for six pollutant categories: NOD = nutrients and oxygen demanding substances, AP = agricultural pesticides, HM = heavy metals, POC = persistent organic compounds, SA = salinity, SSS = suspended solids and sediments.

<u>COLEOPTERA</u>
(as of 10 December 1987)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Chrysomelidae			4	5	3	4	3	4
Chrysomelidae	Donacia		4	5	3	4	3	4
Chrysomelidae	Galerucella		5	5	3	4	4	4
Curculionidae			5	5	3	4	4	4
Dryopidae			2	4	3	3	3	4
Dryopidae	Helichus		2	4	3	3	3	4
Dryopidae	Helichus	basalis	2	4	3	3	3	4
Dryopidae	Helichus	fastigiatus	2	4	3	3	3	4
Dryopidae	Helichus	lithophilus	2	4	3	3	3	4
Dryopidae	Helichus	striatus	1	4	3	3	3	4
Dryopidae	Helichus	suturalis	3	4	3	3	3	4
Dryopidae	Pelonomus		2	5	2	3	3	4
Dryopidae	Pelonomus	obscurus	2	5	2	3	3	4
Dytiscidae			3	5	1	4	3	2
Dytiscidae	Acilius		3	5	1	5	2	2
Dytiscidae	Acilius	fraternus	3	5	1	5	2	2
Dytiscidae	Acilius	semisulcatus	3	5	1	5	2	2
Dytiscidae	Agabus		2	4	2	3	2	1
Dytiscidae	Agabus	ambiguus	1	4	1	3	2	1
Dytiscidae	Agabus	disintegratus	2	4	1	3	2	1
Dytiscidae	Agabus	obliteratus	1	4	1	3	2	1
Dytiscidae	Agabus	semivittatus	3	4	1	3	2	1
Dytiscidae	Agabus	seriatus	1	4	1	3	2	1
Dytiscidae	Agabus	stagninus	3	4	1	3	2	1
Dytiscidae	Bidessus		3	5	1	4	2	2
Dytiscidae	Bidessus	affinis	3	5	1	4	2	2
Dytiscidae	Bidessus	flavicollis	3	5	1	4	2	2
Dytiscidae	Bidessus	lacustris	3	5	1	4	2	2
Dytiscidae	Celina		3	5	1	4	3	2
Dytiscidae	Celina	hubbelli	3	5	1	4	3	2
Dytiscidae	Colymbetes		2	4	1	4	3	1
Dytiscidae	Colymbetes	sculptilis	2	4	1	4	3	1
Dytiscidae	Copelatus		3	4	1	4	3	1
Dytiscidae	Copelatus	chevrolatei	3	4	1	4	3	1
Dytiscidae	Copelatus	glyphicus	3	4	2	4	3	1
Dytiscidae	Coptotomus		2	4	1	4	3	1
Dytiscidae	Coptotomus	interrogatus	2	4	1	4	3	1
Dytiscidae	Coptotomus	longulus	2	4	1	4	3	1
Dytiscidae	Coptotomus	venustus	2	4	1	4	3	1
Dytiscidae	Cybister		3	5	1	5	3	2
Dytiscidae	Cybister	fimbriolatus	3	5	1	5	3	2
Dytiscidae	Desmopachria		3	5	1	4	2	1
Dytiscidae	Desmopachria	convexa	3	5	1	4	2	1
Dytiscidae	Dytiscus		2	4	1	3	3	2
Dytiscidae	Dytiscus	hybridus	2	4	1	3	3	2
Dytiscidae	Eretes		2	4	1	4	3	2
Dytiscidae	Eretes	sticticus	2	4	1	4	3	2

Coleoptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Dytiscidae	Falloporus		2	4	1	3	3	2
Dytiscidae	Falloporus	pilatei	2	4	1	3	3	2
Dytiscidae	Graphoderus		2	4	1	4	3	2
Dytiscidae	Graphoderus	liberus	2	4	1	4	3	2
Dytiscidae	Hydroporus		2	5	1	4	3	2
Dytiscidae	Hydroporus	clypealis	2	5	1	4	3	2
Dytiscidae	Hydroporus	dimidiatus	4	5	1	4	3	2
Dytiscidae	Hydroporus	diversicornis	3	5	1	4	3	2
Dytiscidae	Hydroporus	mixtus	2	5	1	4	3	2
Dytiscidae	Hydroporus	niger	2	5	1	4	3	2
Dytiscidae	Hydroporus	notabilis	1	5	1	4	3	2
Dytiscidae	Hydroporus	ouachitus	1	5	1	4	3	2
Dytiscidae	Hydroporus	rufilabris	1	5	1	4	3	2
Dytiscidae	Hydroporus	shermani	3	5	1	4	3	2
Dytiscidae	Hydroporus	sulphurius	1	5	1	4	3	2
Dytiscidae	Hydroporus	undulatus	3	5	1	4	3	2
Dytiscidae	Hydroporus	vittatipennis	2	5	1	4	3	2
Dytiscidae	Hydroporus	vittatus	2	5	1	4	3	2
Dytiscidae	Hydroporus	wickhami	3	5	1	4	3	2
Dytiscidae	Hydrovatus		2	5	1	4	3	2
Dytiscidae	Hydrovatus	pustulatus	2	5	1	4	3	2
Dytiscidae	Hygrotus		1	5	3	4	3	1
Dytiscidae	Hygrotus	acaroides	2	5	4	4	3	1
Dytiscidae	Hygrotus	dissimilis	1	5	3	4	3	1
Dytiscidae	Hygrotus	impressopunctatus	1	5	3	4	3	1
Dytiscidae	Hygrotus	nubilus	3	5	4	4	3	1
Dytiscidae	Hygrotus	patruelis	1	5	3	4	3	1
Dytiscidae	Hygrotus	sayi	1	5	3	4	3	1
Dytiscidae	Hygrotus	sellatus	1	5	3	4	3	1
Dytiscidae	Illybius		2	5	1	4	3	2
Dytiscidae	Illybius	biguttulus	2	5	1	4	3	2
Dytiscidae	Illybius	laramaeus	2	5	1	4	3	2
Dytiscidae	Illybius	oblitus	1	5	1	4	3	2
Dytiscidae	Laccodytes		3	5	1	4	3	2
Dytiscidae	Laccophilus		3	5	3	4	3	2
Dytiscidae	Laccophilus	fasciatus	3	5	5	4	3	2
Dytiscidae	Laccophilus	maculosus	2	5	3	4	3	2
Dytiscidae	Laccophilus	proximus	3	5	3	4	3	2
Dytiscidae	Laccophilus	quadrilineatus	2	5	3	4	3	2
Dytiscidae	Liodessus		3	5	1	4	3	2
Dytiscidae	Oreodytes		3	5	1	3	3	2
Dytiscidae	Rhantus		2	4	1	4	3	2
Dytiscidae	Rhantus	binotatus	2	4	1	4	3	2
Dytiscidae	Rhantus	gutticollis	2	4	1	4	3	2
Dytiscidae	Thermonectes		2	4	1	3	4	2
Dytiscidae	Thermonectes	basillaris	2	4	1	3	4	2
Dytiscidae	Thermonectes	ornaticollis	2	4	1	3	4	2
Dytiscidae	Uvarus		3	5	1	4	3	2
Elmidae			2	4	1	3	3	3

Coleoptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Elmidae	Ancyronyx		2	4	2	2	2	1
Elmidae	Ancyronyx	variegata	2	4	1	2	2	1
Elmidae	Dubiraphia		3	5	1	3	3	4
Elmidae	Dubiraphia	brevipennis	1	5	1	3	3	4
Elmidae	Dubiraphia	minima	3	5	2	3	3	4
Elmidae	Dubiraphia	vittata	3	5	1	3	3	4
Elmidae	Heterelmis		2	5	1	3	3	2
Elmidae	Heterelmis	vulnerata	2	5	1	3	3	2
Elmidae	Macronychus		2	5	1	3	2	2
Elmidae	Macronychus	glabratus	2	5	1	3	2	2
Elmidae	Microcylloepus		1	3	1	2	2	2
Elmidae	Microcylloepus	pusillus	1	3	1	2	2	2
Elmidae	Optioservus		1	3	1	2	2	1
Elmidae	Optioservus	phaeus	1	3	1	2	1	1
Elmidae	Optioservus	sandersoni	1	3	1	2	2	1
Elmidae	Stenelmis		2	4	1	3	3	4
Elmidae	Stenelmis	beameri	1	4	2	3	3	4
Elmidae	Stenelmis	bicarinata	2	4	1	3	3	4
Elmidae	Stenelmis	crenata	3	4	1	3	3	4
Elmidae	Stenelmis	decorata	2	4	1	3	3	4
Elmidae	Stenelmis	exigua	1	4	1	3	3	4
Elmidae	Stenelmis	lateralis	1	4	1	3	3	4
Elmidae	Stenelmis	sandersoni	2	4	1	3	3	4
Elmidae	Stenelmis	sexlineata	3	4	2	3	3	4
Elmidae	Stenelmis	vittipennis	2	4	1	3	3	4
Gyrinidae			2	4	3	3	3	2
Gyrinidae	Dineutus		2	4	3	3	3	2
Gyrinidae	Dineutus	assimilis	2	4	5	3	3	2
Gyrinidae	Dineutus	carolinus	2	4	3	3	3	2
Gyrinidae	Dineutus	ciliatus	2	4	3	3	3	2
Gyrinidae	Dineutus	productus	2	4	3	3	3	2
Gyrinidae	Dineutus	serrulatus	2	4	3	3	3	2
Gyrinidae	Gyretes		2	4	2	3	3	2
Gyrinidae	Gyrinus		2	4	3	3	3	2
Gyrinidae	Gyrinus	aeneolus	2	4	5	3	3	2
Gyrinidae	Gyrinus	analis	2	4	3	3	3	2
Gyrinidae	Gyrinus	maculiventris	2	4	3	3	3	2
Gyrinidae	Gyrinus	parcus	2	4	3	3	3	2
Haliplidae			3	4	3	3	3	2
Haliplidae	Haliplus		3	4	2	3	3	2
Haliplidae	Haliplus	borealis	3	4	2	3	3	2
Haliplidae	Haliplus	deceptus	3	4	2	3	3	2
Haliplidae	Haliplus	fasciatus	3	4	2	3	3	2
Haliplidae	Haliplus	oklahomensis	3	4	2	3	3	2
Haliplidae	Haliplus	pantherinus	3	4	2	3	3	2
Haliplidae	Haliplus	tortilipenis	3	4	2	3	3	2
Haliplidae	Haliplus	triopsis	3	4	2	3	3	2
Haliplidae	Haliplus	tumidus	3	4	2	3	3	2
Haliplidae	Peltodytes		3	4	3	3	3	3

Coleoptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Haliplidae	Peltodytes	callosus	3	4	3	3	3	3
Haliplidae	Peltodytes	edentulus	3	4	3	3	3	3
Haliplidae	Peltodytes	festivus	3	4	3	3	3	3
Haliplidae	Peltodytes	lengi	3	4	4	3	3	3
Haliplidae	Peltodytes	littoralis	4	4	3	3	4	3
Haliplidae	Peltodytes	sexmaculatus	3	4	3	3	3	3
Helodidae			4	3	4	2	4	3
Helodidae	Cyphon		4	3	4	2	4	3
Helodidae	Prionocyphon		4	4	4	3	4	3
Heteroceridae			4	4	3	3	3	3
Hydraenidae			3	3	2	2	3	3
Hydraenidae	Ochthebius		3	3	2	2	3	3
Hydrophilidae			3	4	1	3	3	3
Hydrophilidae	Berosus		3	4	3	3	3	3
Hydrophilidae	Berosus	exiguus	3	4	3	3	3	3
Hydrophilidae	Berosus	fraternus	3	4	3	3	3	3
Hydrophilidae	Berosus	infuscatus	3	4	5	3	3	3
Hydrophilidae	Berosus	miles	3	4	3	3	3	3
Hydrophilidae	Berosus	pantherinus	3	4	3	3	3	3
Hydrophilidae	Berosus	peregrinus	3	4	3	3	3	3
Hydrophilidae	Berosus	pugnax	3	4	3	3	3	3
Hydrophilidae	Berosus	striatus	3	4	5	3	3	4
Hydrophilidae	Berosus	stylifer	3	4	3	3	3	3
Hydrophilidae	Cercyon		3	3	1	3	3	3
Hydrophilidae	Cercyon	herceus	3	3	1	3	3	3
Hydrophilidae	Cercyon	mendax	3	3	1	3	3	3
Hydrophilidae	Cercyon	praetextatus	3	3	1	3	3	3
Hydrophilidae	Cercyon	quisquilius	3	3	1	3	3	3
Hydrophilidae	Chaetarthria		3	4	1	3	3	3
Hydrophilidae	Chaetarthria	atra	3	4	1	3	3	3
Hydrophilidae	Chaetarthria	atroides	3	4	1	3	3	3
Hydrophilidae	Chaetarthria	pallida	3	4	1	3	3	3
Hydrophilidae	Cryptopleurum		3	4	1	3	2	3
Hydrophilidae	Cryptopleurum	subtile	3	4	1	3	2	3
Hydrophilidae	Cymbiodyta		3	3	1	2	3	3
Hydrophilidae	Cymbiodyta	beckeri	3	3	1	2	3	3
Hydrophilidae	Cymbiodyta	chamberlaini	3	3	1	2	3	3
Hydrophilidae	Cymbiodyta	semistriata	3	3	1	2	3	3
Hydrophilidae	Cymbiodyta	toddi	3	3	1	2	3	3
Hydrophilidae	Cymbiodyta	vindicata	3	3	1	2	3	3
Hydrophilidae	Dibolocelus		3	4	1	3	3	3
Hydrophilidae	Dibolocelus	ovatus	3	4	1	3	3	3
Hydrophilidae	Elophorus		3	4	1	3	3	3
Hydrophilidae	Elophorus	auricollis	3	4	1	3	3	3
Hydrophilidae	Elophorus	frosti	3	4	1	3	3	3
Hydrophilidae	Elophorus	leechi	3	4	1	3	3	3
Hydrophilidae	Elophorus	linearis	3	4	1	3	3	3
Hydrophilidae	Elophorus	lineatus	3	4	1	3	3	3
Hydrophilidae	Enochrus		3	3	2	2	3	2

Coleoptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Hydrophilidae	Enochrus	cinctus	3	3	2	2	3	2
Hydrophilidae	Enochrus	consortus	3	3	2	2	3	2
Hydrophilidae	Enochrus	cristatus	3	3	2	2	3	2
Hydrophilidae	Enochrus	diffusus	3	3	2	2	3	2
Hydrophilidae	Enochrus	hamiltoni	3	3	2	2	3	2
Hydrophilidae	Enochrus	ochraceus	3	3	2	2	3	2
Hydrophilidae	Enochrus	perplexus	3	3	1	2	3	2
Hydrophilidae	Enochrus	pygmaeus	3	3	2	2	3	2
Hydrophilidae	Enochrus	sayi	3	3	2	2	3	2
Hydrophilidae	Epimetopus		3	4	1	3	2	3
Hydrophilidae	Helobata		3	4	1	3	3	3
Hydrophilidae	Helochares		2	4	1	3	3	3
Hydrophilidae	Helochares	maculicollis	2	4	1	3	3	3
Hydrophilidae	Helocombus		3	4	1	3	2	3
Hydrophilidae	Helophorus		3	3	1	2	3	1
Hydrophilidae	Hydrobius		1	4	1	3	3	1
Hydrophilidae	Hydrobius	fuscipes	1	4	1	3	3	1
Hydrophilidae	Hydrochara		3	4	1	3	3	3
Hydrophilidae	Hydrochara	obtusata	3	4	1	3	3	3
Hydrophilidae	Hydrochus		3	3	1	3	3	2
Hydrophilidae	Hydrochus	neosquamifer	3	3	1	3	3	2
Hydrophilidae	Hydrochus	pseudosquamifer	3	3	1	3	3	2
Hydrophilidae	Hydrochus	rufipes	3	3	1	3	3	2
Hydrophilidae	Hydrochus	scabratus	3	3	1	3	3	2
Hydrophilidae	Hydrochus	squamifer	3	3	1	3	3	2
Hydrophilidae	Hydrochus	vagas	3	3	1	3	3	2
Hydrophilidae	Hydrophilus		2	4	1	3	3	3
Hydrophilidae	Hydrophilus	triangularis	2	4	1	3	3	3
Hydrophilidae	Laccobius		2	4	1	3	3	3
Hydrophilidae	Laccobius	carri	2	4	1	3	3	3
Hydrophilidae	Laccobius	ellipticus	2	4	1	3	3	3
Hydrophilidae	Laccobius	magnus	2	4	1	3	3	3
Hydrophilidae	Laccobius	minutoides	2	4	1	3	3	3
Hydrophilidae	Laccobius	reflexipenis	2	4	1	3	3	3
Hydrophilidae	Laccobius	spangleri	2	4	1	3	3	3
Hydrophilidae	Laccobius	teneralis	2	4	1	3	3	3
Hydrophilidae	Paracymus		4	3	1	2	3	4
Hydrophilidae	Paracymus	communis	4	3	1	2	3	4
Hydrophilidae	Paracymus	confusus	4	3	1	2	3	4
Hydrophilidae	Paracymus	despectus	4	3	1	2	3	4
Hydrophilidae	Paracymus	subcupreus	4	3	1	2	3	4
Hydrophilidae	Phaenonotum		3	4	1	3	3	3
Hydrophilidae	Phaenonotum	exstriatum	3	4	1	3	3	3
Hydrophilidae	Sperchopsis		3	4	1	3	3	2
Hydrophilidae	Sperchopsis	tessalatus	3	4	1	3	3	2
Hydrophilidae	Sphaeridium		3	4	1	3	3	3
Hydrophilidae	Sphaeridium	bipustulatum	3	4	1	3	3	3
Hydrophilidae	Tropisternus		3	4	1	3	3	3
Hydrophilidae	Tropisternus	blatchleyi	3	4	1	3	3	3

Coleoptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Hydrophilidae	Tropisternus	cillaris	3	4	1	3	3	3
Hydrophilidae	Tropisternus	columbianus	3	4	1	3	3	3
Hydrophilidae	Tropisternus	ellipticus	3	4	1	3	3	3
Hydrophilidae	Tropisternus	lateralis	3	4	1	3	3	3
Hydrophilidae	Tropisternus	natator	3	4	1	3	3	3
Limnichidae			1	3	1	3	2	2
Limnichidae	Limnichus		1	3	1	3	2	2
Limnichidae	Lutrochus		1	3	1	3	2	2
Limnichidae	Lutrochus	laticeps	1	3	1	3	2	2
Noteridae			3	4	1	4	3	2
Noteridae	Hydrocanthus		3	4	1	4	3	2
Noteridae	Hydrocanthus	similator	3	4	1	4	3	2
Psephenidae			2	3	3	2	1	3
Psephenidae	Ectopria		2	3	0	2	1	2
Psephenidae	Psephenus		2	3	5	2	1	3
Psephenidae	Psephenus	herricki	2	3	5	2	1	3
Staphylinidae			3	4	5	3	5	2

DIPTERA
(as of 30 January 1988)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Anthomyiidae			3	3	1	3	4	4
Ceratopogonidae			3	3	3	3	3	4
Ceratopogonidae	Atrichopogon		3	3	2	3	2	2
Ceratopogonidae	Culicoides		4	3	3	3	3	4
Ceratopogonidae	Forcipomyia		3	3	3	3	3	4
Ceratopogonidae	Jenkinshelen		3	3	4	3	3	4
Ceratopogonidae	Palpomyia		3	3	3	3	3	4
Ceratopogonidae	Probezzia		3	3	3	3	3	4
Ceratopogonidae	Probezzia	pallida	3	3	3	3	3	4
Chaoboridae			4	4	4	3	3	4
Chaoboridae	Chaoborus		4	4	4	3	3	4
Chaoboridae	Chaoborus	americanus	4	4	4	3	3	4
Chaoboridae	Chaoborus	flavicans	4	4	4	3	3	4
Chaoboridae	Chaoborus	punctipennis	4	4	4	3	3	4
Chironomidae			3	3	2	3	3	4
Chironomidae	Ablabesmyia		3	3	1	3	3	3
Chironomidae	Ablabesmyia	annulata	3	3	2	3	3	3
Chironomidae	Ablabesmyia	aurea	3	3	1	3	3	3
Chironomidae	Ablabesmyia	illinoense	2	3	1	3	3	3
Chironomidae	Ablabesmyia	mallochi	3	3	1	3	3	3
Chironomidae	Ablabesmyia	monilis	3	3	2	3	3	3
Chironomidae	Ablabesmyia	peleensis	3	3	1	3	3	3
Chironomidae	Ablabesmyia	pulchripennis	2	3	1	3	2	3
Chironomidae	Antillocladius		3	3	1	3	3	4
Chironomidae	Antillocladius	arcuatus	3	3	1	3	3	4
Chironomidae	Antillocladius	pluspilalus	3	3	1	3	3	4
Chironomidae	Axarus		3	3	1	3	3	4
Chironomidae	Axarus	festivus	3	3	1	3	3	4
Chironomidae	Axarus	scopula	3	3	1	3	3	4
Chironomidae	Axarus	taenionotus	3	3	1	3	3	4
Chironomidae	Boreochlus		3	3	0	3	3	4
Chironomidae	Brillia		1	3	0	2	1	2
Chironomidae	Bryophaenocladius		3	3	0	3	3	4
Chironomidae	Camptocladius		3	3	0	3	3	4
Chironomidae	Camptocladius	stercorarius	3	3	0	3	3	4
Chironomidae	Cardiocladius		3	3	1	3	3	4
Chironomidae	Chaetocladius		2	3	3	3	2	3
Chironomidae	Chernovskiia		3	3	0	3	3	3
Chironomidae	Chernovskiia	amphitrite	3	3	0	3	3	3
Chironomidae	Chernovskiia	orbicus	3	3	0	3	3	3
Chironomidae	Chironomus		5	5	4	3	4	5
Chironomidae	Chironomus	attenuatus	4	5	4	3	4	5
Chironomidae	Chironomus	crassicaudatus	4	5	4	3	4	5
Chironomidae	Chironomus	decorus	5	5	4	3	4	5
Chironomidae	Chironomus	plumosus	5	5	3	3	4	5
Chironomidae	Chironomus	riparius	5	5	5	3	4	5
Chironomidae	Chironomus	staegeri	4	5	3	3	4	5

Diptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Chironomidae	Cladopelma		4	2	1	2	2	4
Chironomidae	Cladopelma	amachaerus	4	2	1	2	2	4
Chironomidae	Cladopelma	collator	4	2	1	2	2	4
Chironomidae	Cladopelma	edwardsi	4	2	1	2	2	4
Chironomidae	Cladopelma	galeator	4	2	1	2	2	4
Chironomidae	Cladopelma	viridula	4	2	1	2	2	4
Chironomidae	Cladotanytarsus		3	3	2	3	2	3
Chironomidae	Clinotanypus		3	2	1	2	3	3
Chironomidae	Clinotanypus	pinguis	3	2	1	2	3	3
Chironomidae	Coelotanypus		2	2	1	2	3	3
Chironomidae	Coelotanypus	atus	2	2	1	2	3	3
Chironomidae	Coelotanypus	concinnus	1	2	1	2	3	3
Chironomidae	Coelotanypus	scapularis	2	2	1	2	3	3
Chironomidae	Coelotanypus	tricolor	2	2	1	2	3	3
Chironomidae	Conchapelopia		3	3	3	3	4	3
Chironomidae	Conchapelopia	aleta	3	3	2	3	4	3
Chironomidae	Conchapelopia	dusena	3	3	3	3	4	3
Chironomidae	Conchapelopia	goniodes	3	3	3	3	4	3
Chironomidae	Conchapelopia	rurika	3	3	3	3	4	3
Chironomidae	Conchapelopia	telema	3	3	3	3	3	3
Chironomidae	Constempellina		3	3	0	3	3	4
Chironomidae	Corynoneura		2	4	5	3	3	3
Chironomidae	Cricotopus		4	3	3	3	3	4
Chironomidae	Cricotopus	absurdus	4	3	1	3	3	4
Chironomidae	Cricotopus	bicinctus	4	3	5	3	4	4
Chironomidae	Cricotopus	exilus	4	3	4	3	3	4
Chironomidae	Cricotopus	infuscatus	4	3	4	3	4	4
Chironomidae	Cricotopus	sylvestris	4	3	1	3	2	4
Chironomidae	Cricotopus	tremulus	4	3	2	3	3	4
Chironomidae	Cricotopus	tricinctus	4	3	3	3	4	4
Chironomidae	Cricotopus	trifascia	4	3	2	3	3	4
Chironomidae	Cricotopus	trifasciatus	4	3	3	3	3	4
Chironomidae	Cryptochironomus		4	3	3	3	3	4
Chironomidae	Cryptochironomus	blarina	4	3	3	3	3	4
Chironomidae	Cryptochironomus	digitatus	4	3	3	3	3	4
Chironomidae	Cryptochironomus	fulvus	4	3	4	3	3	4
Chironomidae	Cryptochironomus	sorex	4	3	3	3	3	4
Chironomidae	Cryptotendipes		3	3	2	3	3	3
Chironomidae	Cryptotendipes	ariel	3	3	2	3	3	3
Chironomidae	Cryptotendipes	casuarius	3	3	2	3	3	3
Chironomidae	Cryptotendipes	emorsus	3	3	2	3	3	3
Chironomidae	Cyphomella		3	2	1	2	3	3
Chironomidae	Cyphomella	cornea	3	2	1	2	3	3
Chironomidae	Diamesa		2	2	1	2	1	3
Chironomidae	Diamesa	chiobates	2	2	1	2	1	3
Chironomidae	Diamesa	hadaki	2	2	1	2	1	3
Chironomidae	Diamesa	heteropus	2	2	0	2	1	3
Chironomidae	Diamesa	nivoriunda	2	2	1	2	1	3
Chironomidae	Dicrotendipes		4	3	3	3	4	4

Diptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Chironomidae	Dicrotendipes	botaurus	4	3	3	3	4	4
Chironomidae	Dicrotendipes	fumidus	4	3	3	3	4	4
Chironomidae	Dicrotendipes	lucifer	4	3	3	3	4	4
Chironomidae	Dicrotendipes	modestus	4	3	2	3	4	4
Chironomidae	Dicrotendipes	nemodestus	4	3	4	3	4	4
Chironomidae	Dicrotendipes	nervosus	4	3	2	3	4	4
Chironomidae	Diplocladius		1	2	1	2	1	2
Chironomidae	Diplocladius	cultriger	1	2	1	2	1	2
Chironomidae	Djalmabatista		3	3	1	3	3	4
Chironomidae	Einfeldia		5	2	3	2	5	5
Chironomidae	Einfeldia	brunneipennis	5	2	3	2	5	5
Chironomidae	Einfeldia	chelonia	5	2	3	2	5	5
Chironomidae	Einfeldia	dorsalis	5	2	3	2	5	5
Chironomidae	Einfeldia	paganus	5	2	3	2	5	5
Chironomidae	Endochironomus		3	3	3	3	3	3
Chironomidae	Endochironomus	nigricans	3	3	3	3	3	3
Chironomidae	Endochironomus	subtendens	3	3	3	3	3	3
Chironomidae	Epoicocladius		2	2	0	2	2	3
Chironomidae	Eukiefferiella		2	3	2	3	3	3
Chironomidae	Eukiefferiella	brevinervis	2	3	4	3	3	3
Chironomidae	Eukiefferiella	claripennis	2	3	2	3	3	3
Chironomidae	Eukiefferiella	ilkeyensis	2	3	2	3	3	3
Chironomidae	Fittkauimyia		3	3	1	3	3	4
Chironomidae	Gillotia		3	2	1	2	3	3
Chironomidae	Gillotia	alboviridis	3	2	1	2	3	3
Chironomidae	Glyptotendipes		5	3	1	3	4	5
Chironomidae	Glyptotendipes	barbipes	5	3	1	3	4	5
Chironomidae	Glyptotendipes	lobiferus	5	3	1	3	4	5
Chironomidae	Glyptotendipes	paripes	5	3	1	3	4	5
Chironomidae	Goeldichironomus		5	2	2	2	4	4
Chironomidae	Goeldichironomus	holoprasinus	5	2	2	2	4	4
Chironomidae	Gymnometriocnemus		3	3	0	3	3	4
Chironomidae	Harnischia		4	3	1	3	3	4
Chironomidae	Harnischia	curtilamellata	4	3	1	3	3	4
Chironomidae	Harnischia	incidata	4	3	1	3	3	4
Chironomidae	Hayesomyia		3	3	1	3	3	3
Chironomidae	Hayesomyia	senata	3	3	1	3	3	3
Chironomidae	Heleniella		3	3	0	3	3	4
Chironomidae	Heleniella	parva	3	3	0	3	3	4
Chironomidae	Helopelopia		3	3	2	3	3	3
Chironomidae	Helopelopia	cornuticaudata	3	3	2	3	3	3
Chironomidae	Heterotrissocladius		2	2	2	2	1	3
Chironomidae	Hydrobaenus		2	2	1	2	1	4
Chironomidae	Hydrobaenus	johannseni	2	2	1	2	1	4
Chironomidae	Hydrobaenus	pilipes	2	2	1	2	1	4
Chironomidae	Hydrobaenus	pilopodex	2	2	1	2	1	4
Chironomidae	Kiefferulus		4	2	1	2	3	4
Chironomidae	Kiefferulus	dux	4	2	1	2	3	4
Chironomidae	Krenosmittia		3	3	0	3	3	4

Diptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Chironomidae	Labrundinia		2	3	3	3	2	3
Chironomidae	Labrundinia	maculata	2	3	4	3	2	3
Chironomidae	Labrundinia	neopilosella	2	3	3	3	2	3
Chironomidae	Labrundinia	pillosella	2	3	3	3	2	3
Chironomidae	Larsia		3	3	3	3	3	3
Chironomidae	Larsia	arcuata	3	3	3	3	3	3
Chironomidae	Larsia	decolorata	3	3	3	3	3	3
Chironomidae	Larsia	indistincta	3	3	3	3	3	3
Chironomidae	Larsia	lyra	3	3	3	3	3	3
Chironomidae	Larsia	marginella	3	3	3	3	3	3
Chironomidae	Larsia	pallens	3	3	3	3	3	3
Chironomidae	Larsia	planesis	3	3	3	3	3	3
Chironomidae	Lauterborniella		3	2	1	2	3	3
Chironomidae	Lauterborniella	agrayloides	3	2	1	2	3	3
Chironomidae	Lenziella		3	2	1	2	3	3
Chironomidae	Lenziella	cruscula	3	2	1	2	3	3
Chironomidae	Limnophyes		3	2	3	2	3	3
Chironomidae	Limnophyes	cristatissimus	3	2	1	2	3	3
Chironomidae	Limnophyes	hudsoni	3	2	4	2	3	3
Chironomidae	Lopescladius		3	3	1	3	3	4
Chironomidae	Lopescladius	inermis	3	3	1	3	3	4
Chironomidae	Meropelopia		3	3	2	3	3	3
Chironomidae	Meropelopia	americana	3	3	2	3	3	3
Chironomidae	Meropelopia	flavifrons	3	3	2	3	3	3
Chironomidae	Mesosmittia		3	3	0	3	3	4
Chironomidae	Mesosmittia	patrihortae	3	3	0	3	3	4
Chironomidae	Mesosmittia	prolixa	3	3	0	3	3	4
Chironomidae	Metriocnemus		2	2	1	2	3	4
Chironomidae	Microchironomus		4	2	1	2	3	4
Chironomidae	Microchironomus	nigrovittatus	4	2	1	2	3	4
Chironomidae	Micropsectra		3	2	2	2	3	3
Chironomidae	Micropsectra	nigripila	3	2	2	2	3	3
Chironomidae	Microtendipes		3	3	1	3	3	3
Chironomidae	Microtendipes	pedullus	3	3	1	3	3	3
Chironomidae	Nanocladius		1	2	1	2	2	3
Chironomidae	Nanocladius	anderseni	2	2	1	2	2	3
Chironomidae	Nanocladius	balticus	1	2	1	2	2	3
Chironomidae	Nanocladius	crassicornis	2	2	1	2	2	3
Chironomidae	Nanocladius	distinctus	1	2	1	2	2	3
Chironomidae	Nanocladius	incomptus	1	2	1	2	2	3
Chironomidae	Nanocladius	minimus	2	2	1	2	2	3
Chironomidae	Nanocladius	spiniplenus	1	2	1	2	2	3
Chironomidae	Natarsia		3	3	3	3	3	3
Chironomidae	Natarsia	baltimoreus	3	3	3	3	3	3
Chironomidae	Neozavrelia		3	3	0	3	3	4
Chironomidae	Nilotanypus		3	3	1	3	2	3
Chironomidae	Nilotanypus	fimbriatus	3	3	1	3	2	3
Chironomidae	Nimbocera		3	3	1	3	3	4
Chironomidae	Nimbocera	kansensis	3	3	1	3	3	4

Diptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Chironomidae	Orthocladius		3	2	1	2	3	3
Chironomidae	Orthocladius	abiskoensis	3	2	1	2	3	3
Chironomidae	Orthocladius	carlatus	3	2	1	2	3	3
Chironomidae	Orthocladius	dorenus	3	2	1	2	3	3
Chironomidae	Orthocladius	ferringtoni	3	2	1	2	3	3
Chironomidae	Orthocladius	mallochi	3	2	1	2	3	3
Chironomidae	Orthocladius	obumbratus	3	2	2	2	3	3
Chironomidae	Orthocladius	rivicola	3	2	1	2	3	3
Chironomidae	Orthocladius	rivulorum	3	2	1	2	3	3
Chironomidae	Orthocladius	thienemanni	3	2	2	2	3	3
Chironomidae	Paraboreochlus		3	3	0	3	3	4
Chironomidae	Parachaetocladius		3	3	1	3	3	4
Chironomidae	Parachaetocladius	hudsoni	3	3	1	3	3	4
Chironomidae	Parachironomus		4	3	1	3	3	4
Chironomidae	Parachironomus	abortivus	4	3	1	3	3	4
Chironomidae	Parachironomus	carinatus	4	3	1	3	3	4
Chironomidae	Parachironomus	chaetaolus	4	3	1	3	3	4
Chironomidae	Parachironomus	directus	4	3	1	3	3	4
Chironomidae	Parachironomus	frequens	4	3	1	3	3	4
Chironomidae	Parachironomus	monochromus	4	3	1	3	3	4
Chironomidae	Parachironomus	potamogeti	4	3	1	3	3	4
Chironomidae	Parachironomus	tenuicaudatus	4	3	1	3	3	4
Chironomidae	Paracladopelma		3	3	1	3	3	3
Chironomidae	Paracladopelma	doris	3	3	1	3	3	3
Chironomidae	Paracladopelma	longanae	3	3	1	3	3	3
Chironomidae	Paracladopelma	nereis	3	3	1	3	3	3
Chironomidae	Paracladopelma	undine	3	3	1	3	3	3
Chironomidae	Paracricotopus		3	3	0	3	3	4
Chironomidae	Parakiefferiella		2	2	3	2	3	3
Chironomidae	Parakiefferiella	coronata	2	2	5	2	3	3
Chironomidae	Paralauterborniella		3	3	1	3	3	3
Chironomidae	Paralauterborniella	elachista	3	3	1	3	3	3
Chironomidae	Paralauterborniella	nigrohalteralis	3	3	1	3	3	3
Chironomidae	Paralauterborniella	subcincta	3	3	1	3	3	3
Chironomidae	Paramerina		2	3	1	3	3	3
Chironomidae	Paramerina	smithae	2	3	1	3	3	3
Chironomidae	Parametriocnemus		3	2	1	2	3	3
Chironomidae	Parametriocnemus	lundbecki	3	2	1	2	3	3
Chironomidae	Paraphaenocladius		2	2	1	2	2	3
Chironomidae	Paraphaenocladius	exagitans	2	2	1	2	2	3
Chironomidae	Paratanytarsus		3	2	3	2	3	3
Chironomidae	Paratendipes		3	2	1	2	3	3
Chironomidae	Paratendipes	albimanus	3	2	1	2	3	3
Chironomidae	Paratendipes	basidens	3	2	1	2	3	3
Chironomidae	Paratendipes	nitidulus	3	2	1	2	3	3
Chironomidae	Paratendipes	subaequalis	3	3	1	3	3	4
Chironomidae	Pentaneura		2	3	3	3	2	3
Chironomidae	Pentaneura	inconspicua	2	3	3	3	2	3
Chironomidae	Pentaneura	inyoensis	2	3	4	3	2	3

Diptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Chironomidae	Phaenopsectra		4	2	1	2	3	4
Chironomidae	Phaenopsectra	flavipes	4	2	1	2	3	4
Chironomidae	Phaenopsectra	punctipes	4	2	1	2	3	4
Chironomidae	Polypedilum		3	3	3	3	3	3
Chironomidae	Polypedilum	apicatum	3	3	3	3	3	3
Chironomidae	Polypedilum	aviceps	3	3	3	3	3	3
Chironomidae	Polypedilum	braseniae	3	3	3	3	3	3
Chironomidae	Polypedilum	convictum	3	3	5	3	3	3
Chironomidae	Polypedilum	digitifer	3	3	3	3	3	3
Chironomidae	Polypedilum	fallax	3	3	3	3	3	3
Chironomidae	Polypedilum	floridense	3	3	3	3	3	3
Chironomidae	Polypedilum	griseopunctatum	3	3	3	3	3	3
Chironomidae	Polypedilum	halterale	3	3	2	3	3	3
Chironomidae	Polypedilum	illinoense	3	3	3	3	3	3
Chironomidae	Polypedilum	nubeculosum	3	3	3	3	3	3
Chironomidae	Polypedilum	ontario	3	3	3	3	3	3
Chironomidae	Polypedilum	pedatum	3	3	3	3	3	3
Chironomidae	Polypedilum	scalaenum	3	3	2	3	3	3
Chironomidae	Polypedilum	simulans	3	3	3	3	3	3
Chironomidae	Polypedilum	sordens	3	3	2	3	3	3
Chironomidae	Polypedilum	trigonus	3	3	3	3	3	3
Chironomidae	Polypedilum	tritum	3	3	2	3	3	3
Chironomidae	Potthastia		2	2	1	2	1	3
Chironomidae	Potthastia	gaedii	2	2	1	2	1	3
Chironomidae	Procladius		3	3	3	3	4	4
Chironomidae	Procladius	bellus	4	3	3	3	4	4
Chironomidae	Procladius	culiciformis	4	3	4	3	4	4
Chironomidae	Procladius	freemani	3	3	3	3	4	4
Chironomidae	Procladius	riparius	3	3	3	3	4	4
Chironomidae	Procladius	sublettei	3	3	3	3	4	4
Chironomidae	Psectrocladius		2	2	5	2	3	3
Chironomidae	Psectrocladius	vernalis	2	2	3	2	3	3
Chironomidae	Psectrotanypus		3	3	3	3	3	3
Chironomidae	Psectrotanypus	dyari	4	3	3	3	3	3
Chironomidae	Pseudochironomus		3	2	1	2	3	3
Chironomidae	Pseudochironomus	aureus	3	2	1	2	3	3
Chironomidae	Pseudochironomus	fulviventris	3	2	1	2	3	3
Chironomidae	Pseudochironomus	pseudoviridis	3	2	1	2	3	3
Chironomidae	Pseudochironomus	richardsoni	3	2	1	2	3	3
Chironomidae	Pseudorthocladius		3	3	1	3	3	4
Chironomidae	Pseudosmittia		1	2	0	2	2	3
Chironomidae	Pseudosmittia	forcipata	1	2	3	2	2	3
Chironomidae	Psilometriocnemus		3	3	1	3	3	4
Chironomidae	Psilometriocnemus	triannulatus	3	3	1	3	3	4
Chironomidae	Rheocricotopus		3	3	1	3	2	3
Chironomidae	Rheosmittia		3	3	0	3	3	4
Chironomidae	Rheotanytarsus		3	2	1	2	3	3
Chironomidae	Rheotanytarsus	akrina	3	2	1	2	3	3
Chironomidae	Rheotanytarsus	exiguus	3	2	1	2	3	3

<u>Diptera (continued)</u>

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Chironomidae	Robackia		3	2	1	2	3	3
Chironomidae	Robackia	claviger	3	2	1	2	3	3
Chironomidae	Saetheria		3	2	1	2	3	3
Chironomidae	Saetheria	tylus	3	2	1	2	3	3
Chironomidae	Smittia		3	3	1	3	3	4
Chironomidae	Smittia	aterrima	3	3	1	3	3	4
Chironomidae	Stelechomyia		3	3	1	3	3	4
Chironomidae	Stelechomyia	perpulchra	3	3	1	3	3	4
Chironomidae	Stempellina		2	2	0	2	2	3
Chironomidae	Stempellinella		3	3	1	3	3	4
Chironomidae	Stenochironomus		2	3	0	3	2	3
Chironomidae	Stenochironomus	cinctus	2	3	0	3	2	3
Chironomidae	Stenochironomus	hilaris	2	3	0	3	2	3
Chironomidae	Stenochironomus	macateei	2	3	0	3	2	3
Chironomidae	Stenochironomus	unictus	2	3	0	3	2	3
Chironomidae	Stictochironomus		3	2	3	2	3	3
Chironomidae	Stictochironomus	albricus	3	2	1	2	3	3
Chironomidae	Stictochironomus	annulicrus	3	2	1	2	3	3
Chironomidae	Stictochironomus	naevus	3	2	1	2	3	3
Chironomidae	Stictochironomus	palliatus	2	3	0	3	2	3
Chironomidae	Stictochironomus	varius	3	2	1	2	3	3
Chironomidae	Stilocladius		3	3	0	3	3	4
Chironomidae	Sympotthastia		2	2	1	2	1	3
Chironomidae	Tanypus		4	2	1	2	3	4
Chironomidae	Tanypus	concavus	4	2	1	2	3	4
Chironomidae	Tanypus	grodhausi	3	2	1	2	3	4
Chironomidae	Tanypus	neopunctipennis	4	2	1	2	3	4
Chironomidae	Tanypus	nubifer	4	2	1	2	3	4
Chironomidae	Tanypus	punctipennis	4	2	1	2	3	4
Chironomidae	Tanypus	stellatus	4	2	1	2	3	4
Chironomidae	Tanytarsus		3	4	3	4	3	3
Chironomidae	Telopelopia		3	3	1	3	3	3
Chironomidae	Telopelopia	okoboji	3	3	1	3	3	3
Chironomidae	Thienemanniella		2	3	3	3	3	3
Chironomidae	Thienemannimyia		3	3	1	3	3	3
Chironomidae	Thienemannimyia	barberi	3	3	1	3	3	3
Chironomidae	Thienemannimyia	norena	3	3	1	3	3	3
Chironomidae	Tribelos		3	2	2	2	3	3
Chironomidae	Tribelos	fuscicornis	3	2	2	2	3	3
Chironomidae	Tribelos	jucundum	3	2	2	2	3	3
Chironomidae	Tvetenia		3	3	2	3	3	4
Chironomidae	Tvetenia	paucunca	3	3	2	3	3	4
Chironomidae	Tvetenia	vitracies	3	3	2	3	3	4
Chironomidae	Xenochironomus		3	2	1	2	4	3
Chironomidae	Xenochironomus	xenolabis	3	2	1	2	4	3
Chironomidae	Zavrelia		3	3	0	3	3	4
Chironomidae	Zavreliella		3	3	0	3	3	4
Chironomidae	Zavreliella	varipennis	3	3	0	3	3	4
Chironomidae	Zavrelimyia		4	3	1	3	3	4

Diptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Chironomidae	Zavrelimyia	sinuosa	3	3	1	3	3	4
Chironomidae	Zavrelimyia	thryptica	4	3	1	3	3	4
Culicidae			4	4	3	4	4	2
Culicidae	Aedes		4	4	3	4	4	2
Culicidae	Aedes	aegypti	4	4	3	4	4	2
Culicidae	Aedes	atlanticus	4	4	3	4	4	2
Culicidae	Aedes	atropalpus	4	4	3	4	4	2
Culicidae	Aedes	canadensis	4	4	3	4	4	2
Culicidae	Aedes	cinerus	4	4	3	4	4	2
Culicidae	Aedes	dorsalis	4	4	3	4	4	2
Culicidae	Aedes	dupreei	4	4	3	4	4	2
Culicidae	Aedes	flaviscens	4	4	3	4	4	2
Culicidae	Aedes	mitchelli	4	4	3	4	4	2
Culicidae	Aedes	nigromaculis	4	4	3	4	4	2
Culicidae	Aedes	sollicitans	4	4	3	4	4	2
Culicidae	Aedes	spenceri	4	4	3	4	4	2
Culicidae	Aedes	stimulans	4	4	3	4	4	2
Culicidae	Aedes	stricticus	4	4	3	4	4	2
Culicidae	Aedes	taeniorhynchus	4	4	3	4	4	2
Culicidae	Aedes	triseriatus	4	4	3	4	4	2
Culicidae	Aedes	trivittatus	4	4	3	4	4	2
Culicidae	Aedes	vexans	4	4	3	4	4	2
Culicidae	Aedes	zoosophus	4	4	3	4	4	2
Culicidae	Anopheles		5	4	3	4	4	2
Culicidae	Anopheles	barberi	5	4	3	4	4	2
Culicidae	Anopheles	crucians	5	4	3	4	4	2
Culicidae	Anopheles	earlei	5	4	3	4	4	2
Culicidae	Anopheles	franciscanus	5	4	3	4	4	2
Culicidae	Anopheles	pseudopunctipennis	5	4	3	4	4	2
Culicidae	Anopheles	punctipennis	5	4	3	4	4	2
Culicidae	Anopheles	quadrimaculatus	5	4	3	4	4	2
Culicidae	Anopheles	walkeri	5	4	3	4	4	2
Culicidae	Coquillettidia		4	4	3	4	4	2
Culicidae	Coquillettidia	perturbans	4	4	3	4	4	2
Culicidae	Culex		5	4	4	4	4	2
Culicidae	Culex	erraticus	5	4	5	4	4	2
Culicidae	Culex	peccator	5	4	3	4	4	2
Culicidae	Culex	pipiens	5	4	4	4	4	2
Culicidae	Culex	quinquefasciatus	5	4	3	4	4	2
Culicidae	Culex	restuans	5	4	4	4	4	2
Culicidae	Culex	salinarius	5	4	3	4	4	2
Culicidae	Culex	tarsalis	5	4	3	4	4	2
Culicidae	Culex	territans	5	4	3	4	4	2
Culicidae	Culiseta		4	4	3	4	4	2
Culicidae	Culiseta	inornata	4	4	3	4	4	2
Culicidae	Culiseta	melanura	4	4	3	4	4	2
Culicidae	Orthopodomyia		4	4	3	4	4	2
Culicidae	Orthopodomyia	alba	4	4	3	4	4	2
Culicidae	Orthopodomyia	signifera	4	4	3	4	4	2

Diptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Culicidae	Psorophora		4	4	3	4	4	2
Culicidae	Psorophora	ciliata	4	4	3	4	4	2
Culicidae	Psorophora	confinnis	4	4	3	4	4	2
Culicidae	Psorophora	cyanescens	4	4	3	4	4	2
Culicidae	Psorophora	datcolor	4	4	3	4	4	2
Culicidae	Psorophora	discolor	4	4	3	4	4	2
Culicidae	Psorophora	ferox	4	4	3	4	4	2
Culicidae	Psorophora	horrida	4	4	3	4	4	2
Culicidae	Psorophora	howardi	4	4	3	4	4	2
Culicidae	Psorophora	longipalpis	4	4	3	4	4	2
Culicidae	Psorophora	signipennis	4	4	3	4	4	2
Culicidae	Toxorhynchites		4	4	3	4	4	2
Culicidae	Toxorhynchites	rutilis	4	4	3	4	4	2
Culicidae	Uranotaenia		4	4	3	4	4	2
Culicidae	Uranotaenia	sappharina	4	4	3	4	4	2
Dolichopodidae			2	3	3	3	3	2
Empididae			3	4	5	3	4	2
Empididae	Hemerodromia		3	4	5	3	4	2
Ephydridae			3	2	4	2	5	3
Ephydridae	Brachydeutera		3	2	4	2	5	3
Ptychopteridae			3	2	3	2	3	3
Ptychopteridae	Bittacomorpha		3	2	3	2	3	3
Ptychopteridae	Bittacomorpha	clavipes	3	2	3	2	3	3
Rhagionidae			2	4	3	3	3	2
Rhagionidae	Atherix		2	4	3	3	3	2
Sciomyzidae			3	4	3	3	3	2
Simuliidae			3	4	2	3	3	2
Simuliidae	Cnephia		1	4	0	3	2	2
Simuliidae	Cnephia	abditoides	1	4	0	3	2	2
Simuliidae	Cnephia	dacotensis	1	4	0	3	2	2
Simuliidae	Simulium		3	4	2	3	3	2
Simuliidae	Simulium	decorum	3	4	2	3	3	2
Simuliidae	Simulium	jenningsi	2	4	2	3	3	2
Simuliidae	Simulium	luggeri	1	4	2	3	3	2
Simuliidae	Simulium	tuberosum	3	4	2	3	3	2
Simuliidae	Simulium	venestum	3	4	2	3	3	2
Simuliidae	Simulium	vittatum	4	4	3	3	3	2
Stratiomyidae			4	3	2	3	4	4
Stratiomyidae	Nemotelus		4	3	2	3	4	4
Stratiomyidae	Odontomyia		4	3	2	3	4	4
Stratiomyidae	Stratiomys		4	2	2	2	4	4
Syrphidae			5	2	3	2	5	4
Syrphidae	Eristalis		5	2	3	2	5	4
Tabanidae			3	3	5	3	5	3
Tabanidae	Chrysops		3	2	4	2	5	4
Tabanidae	Tabanus		3	3	5	3	5	3
Tipulidae			3	3	2	3	3	3
Tipulidae	Antocha		3	2	2	2	3	2
Tipulidae	Antocha	obtusa	3	2	2	2	3	2

Diptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Tipulidae	Cladura		2	3	2	3	3	3
Tipulidae	Cladura	flavoferruginea	2	3	2	3	3	3
Tipulidae	Dactylolabis		2	2	2	2	3	2
Tipulidae	Dactylolabis	montana	2	2	2	2	3	2
Tipulidae	Dicranota		2	3	2	3	3	3
Tipulidae	Empedomorpha		2	3	2	3	3	3
Tipulidae	Empedomorpha	empedoides	2	3	2	3	3	3
Tipulidae	Epiphragma		2	3	2	3	3	3
Tipulidae	Epiphragma	fasciapennis	2	3	2	3	3	3
Tipulidae	Erioptera		3	3	2	3	3	3
Tipulidae	Erioptera	armata	3	3	2	3	3	3
Tipulidae	Erioptera	armillaris	3	3	2	3	3	3
Tipulidae	Erioptera	caliptera	3	3	2	3	3	3
Tipulidae	Erioptera	cana	3	3	2	3	3	3
Tipulidae	Erioptera	cholorphylloides	3	3	2	3	3	3
Tipulidae	Erioptera	furcifer	3	3	2	3	3	3
Tipulidae	Erioptera	graphica	3	3	2	3	3	3
Tipulidae	Erioptera	indianensis	3	3	2	3	3	3
Tipulidae	Erioptera	knabi	3	3	2	3	3	3
Tipulidae	Erioptera	needhami	3	3	2	3	3	3
Tipulidae	Erioptera	parva	3	3	2	3	3	3
Tipulidae	Erioptera	pilipes	3	3	2	3	3	3
Tipulidae	Erioptera	septemtrionis	3	3	2	3	3	3
Tipulidae	Erioptera	straminea	3	3	2	3	3	3
Tipulidae	Erioptera	tantilla	3	3	2	3	3	3
Tipulidae	Erioptera	venusta	3	3	2	3	3	3
Tipulidae	Erioptera	vespertina	3	3	2	3	3	3
Tipulidae	Eugnophomyia		3	3	2	3	3	3
Tipulidae	Eugnophomyia	luctosa	3	3	2	3	3	3
Tipulidae	Gnophomyia		3	3	2	3	3	3
Tipulidae	Gnophomyia	tristissima	3	3	2	3	3	3
Tipulidae	Gonomyia		3	3	2	3	3	3
Tipulidae	Gonomyia	alexanderi	4	3	2	3	3	3
Tipulidae	Gonomyia	blanda	3	3	2	3	3	3
Tipulidae	Gonomyia	cognatella	3	3	2	3	3	3
Tipulidae	Gonomyia	florens	3	3	2	3	3	3
Tipulidae	Gonomyia	gaigei	3	3	2	3	3	3
Tipulidae	Gonomyia	helophila	4	3	2	3	3	3
Tipulidae	Gonomyia	kansensis	4	3	2	3	3	3
Tipulidae	Gonomyia	knowltoniana	3	3	2	3	3	3
Tipulidae	Gonomyia	manca	1	3	2	3	3	3
Tipulidae	Gonomyia	mathesoni	3	3	2	3	3	3
Tipulidae	Gonomyia	slossonae	3	3	2	3	3	3
Tipulidae	Gonomyia	subcinerea	3	3	2	3	3	3
Tipulidae	Gonomyia	sulphurella	4	3	2	3	3	3
Tipulidae	Helius		3	2	2	2	3	3
Tipulidae	Helius	flavipes	3	2	2	2	3	3
Tipulidae	Helius	mainensis	3	2	2	2	3	3
Tipulidae	Hexatoma		3	3	2	3	3	2

Diptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Tipulidae	Hexatoma	brevicornis	3	3	2	3	3	2
Tipulidae	Hexatoma	longicornis	3	3	2	3	3	2
Tipulidae	Limnophila		2	2	2	2	3	3
Tipulidae	Limnophila	auripennis	2	2	2	2	3	3
Tipulidae	Limnophila	fuscovaria	2	2	2	2	3	3
Tipulidae	Limonia		2	3	1	3	3	3
Tipulidae	Limonia	brevivena	2	3	1	3	3	3
Tipulidae	Limonia	canadensis	2	3	1	3	3	3
Tipulidae	Limonia	communis	2	3	1	3	3	2
Tipulidae	Limonia	diversa	2	3	1	3	3	3
Tipulidae	Limonia	divisa	2	3	1	3	3	3
Tipulidae	Limonia	domestica	2	3	1	3	3	3
Tipulidae	Limonia	fallax	2	3	1	3	3	3
Tipulidae	Limonia	globithorax	2	3	1	3	3	3
Tipulidae	Limonia	haeretica	2	3	1	3	3	3
Tipulidae	Limonia	humidicola	2	3	1	3	3	3
Tipulidae	Limonia	immodestoides	2	3	1	3	3	3
Tipulidae	Limonia	intermedia	2	3	1	3	3	3
Tipulidae	Limonia	iowensis	2	3	1	3	3	3
Tipulidae	Limonia	liberta	2	3	1	3	3	3
Tipulidae	Limonia	longipennis	2	3	1	3	3	3
Tipulidae	Limonia	pudica	2	3	1	3	3	3
Tipulidae	Limonia	rara	2	3	1	3	3	3
Tipulidae	Limonia	rostrata	2	3	1	3	3	3
Tipulidae	Limonia	stulta	2	3	1	3	3	3
Tipulidae	Molophilus		3	3	2	3	3	3
Tipulidae	Molophilus	hirtipennis	3	3	2	3	3	3
Tipulidae	Molophilus	pubipennis	3	3	2	3	3	3
Tipulidae	Ormosia		2	3	2	3	3	3
Tipulidae	Ormosia	arculata	2	3	2	3	3	3
Tipulidae	Ormosia	frisoni	2	3	2	3	3	3
Tipulidae	Ormosia	ingloria	2	3	2	3	3	3
Tipulidae	Ormosia	romanovichiana	2	3	2	3	3	3
Tipulidae	Paradelphomyia		3	2	2	2	3	2
Tipulidae	Paradelphomyia	cayuga	3	2	2	2	3	2
Tipulidae	Pedicia		2	3	2	3	3	3
Tipulidae	Pedicia	albivitta	2	3	2	3	3	3
Tipulidae	Pedicia	inconstans	2	3	2	3	3	3
Tipulidae	Pilaria		3	2	2	2	3	2
Tipulidae	Pilaria	imbecilla	3	2	2	2	3	2
Tipulidae	Pilaria	quadrata	3	2	2	2	3	2
Tipulidae	Pilaria	tenuipes	3	2	2	2	3	2
Tipulidae	Pseudolimnophila		1	2	2	2	3	2
Tipulidae	Pseudolimnophila	contempta	1	2	2	2	3	2
Tipulidae	Pseudolimnophila	luteipennis	1	2	2	2	3	2
Tipulidae	Tasiocera		2	3	2	3	3	2
Tipulidae	Tasiocera	ursina	2	3	2	3	3	2
Tipulidae	Teucholabis		3	3	2	3	3	3
Tipulidae	Teucholabis	complexa	3	3	2	3	3	3

Diptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Tipulidae	Teucholabis	immaculata	3	3	2	3	3	3
Tipulidae	Teucholabis	lucida	3	3	2	3	3	3
Tipulidae	Tipula		3	3	2	3	3	3
Tipulidae	Tipula	abdominalis	3	3	2	3	3	3
Tipulidae	Tipula	albimacula	0	3	2	3	3	3
Tipulidae	Tipula	borealis	3	3	2	3	3	3
Tipulidae	Tipula	caloptera	0	3	2	3	3	3
Tipulidae	Tipula	concava	3	3	2	3	3	3
Tipulidae	Tipula	cunctans	2	3	2	3	3	3
Tipulidae	Tipula	dorsimacula	3	3	2	3	3	3
Tipulidae	Tipula	furca	3	3	2	3	3	3
Tipulidae	Tipula	hermannia	3	3	2	3	3	3
Tipulidae	Tipula	ignobilis	2	3	2	3	3	3
Tipulidae	Tipula	illustris	1	3	2	3	3	3
Tipulidae	Tipula	kennicotti	1	3	2	3	3	3
Tipulidae	Tipula	paterifera	2	3	2	3	3	3
Tipulidae	Tipula	sayi	2	3	2	3	3	3
Tipulidae	Tipula	strepens	3	3	2	3	3	3
Tipulidae	Tipula	tricolor	3	3	2	3	3	3
Tipulidae	Tipula	ultima	3	3	2	3	3	3
Tipulidae	Tipula	vicina	2	3	2	3	3	3
Tipulidae	Toxorhina		2	3	2	3	3	3
Tipulidae	Toxorhina	magna	2	3	2	3	3	3

<u>EPHEMEROPTERA</u>
(as of 10 December 1987)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Baetidae			2	0	2	0	3	3
Baetidae	Apobaetis		3	0	3	0	3	3
Baetidae	Baetis		2	0	2	0	3	3
Baetidae	Baetis	dardanus	2	0	2	0	3	3
Baetidae	Baetis	ephippiatus	2	0	2	0	3	3
Baetidae	Baetis	flavistriga	3	0	2	0	3	3
Baetidae	Baetis	intercalaris	3	0	3	0	4	3
Baetidae	Baetis	longipalpus	3	0	2	0	3	3
Baetidae	Baetis	propinquus	2	0	2	0	3	3
Baetidae	Baetis	pygmaeus	2	0	2	0	3	3
Baetidae	Baetis	quilleri	3	0	2	0	3	3
Baetidae	Callibaetis		3	1	3	1	4	2
Baetidae	Centroptilum		1	0	1	0	2	1
Baetidae	Cloeon		2	1	2	1	2	2
Baetidae	Dactylobaetis		2	1	2	1	3	2
Baetidae	Paracloeodes		3	1	2	1	3	2
Baetidae	Pseudocloeon		2	1	2	1	2	2
Caenidae			3	0	3	0	3	3
Caenidae	Brachycercus		2	1	3	1	3	3
Caenidae	Brachycercus	flavus	2	1	3	1	3	3
Caenidae	Brachycercus	lacustris	3	1	3	1	3	3
Caenidae	Caenis		3	0	2	0	3	3
Caenidae	Caenis	delicata	4	0	3	0	3	3
Caenidae	Caenis	hilaris	2	0	2	0	2	3
Caenidae	Caenis	jacosa	4	0	3	0	3	3
Caenidae	Caenis	punctata	1	0	1	0	1	3
Caenidae	Caenis	ridens	3	0	2	0	3	3
Caenidae	Caenis	simulans	4	0	3	0	4	4
Ephemerellidae			1	1	1	1	3	3
Ephemerellidae	Ephemerella		1	1	1	1	3	3
Ephemerellidae	Eurylophella		2	1	2	1	2	3
Ephemeridae			2	2	2	2	2	3
Ephemeridae	Ephemera		1	0	1	0	2	3
Ephemeridae	Ephemera	simulans	1	0	1	0	2	3
Ephemeridae	Hexagenia		3	3	3	3	3	3
Ephemeridae	Hexagenia	atrocaudata	2	3	2	3	2	2
Ephemeridae	Hexagenia	bilineata	3	3	3	3	3	4
Ephemeridae	Hexagenia	limbata	3	3	3	3	3	2
Ephemeridae	Hexagenia	rigida	3	3	2	3	2	3
Heptageniidae			2	2	2	2	3	3
Heptageniidae	Anepeorus		2	2	2	2	3	3
Heptageniidae	Heptagenia		2	2	2	2	3	3
Heptageniidae	Heptagenia	diabasia	3	2	2	2	3	3
Heptageniidae	Heptagenia	flavescens	2	2	2	2	3	3
Heptageniidae	Heptagenia	maculipennis	2	2	2	2	3	4
Heptageniidae	Heptagenia	marginalis	2	2	2	2	3	4
Heptageniidae	Heptagenia	pulla	0	2	2	2	3	3

Ephemeroptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Heptageniidae	Macdunnoa		2	2	2	2	3	3
Heptageniidae	Pseudiron		3	2	2	2	3	3
Heptageniidae	Pseudiron	centralis	3	2	2	2	3	3
Heptageniidae	Stenacron		4	2	3	2	3	3
Heptageniidae	Stenacron	interpunctatum	4	2	3	2	3	3
Heptageniidae	Stenonema		2	2	2	2	3	4
Heptageniidae	Stenonema	exiguum	2	2	2	2	3	3
Heptageniidae	Stenonema	femoratum	3	2	3	2	3	4
Heptageniidae	Stenonema	integrum	3	2	2	2	3	4
Heptageniidae	Stenonema	mediopunctatum	2	2	2	2	2	4
Heptageniidae	Stenonema	pulchellum	2	2	2	2	3	4
Heptageniidae	Stenonema	terminatum	2	2	2	2	3	4
Leptophlebiidae			2	1	2	1	2	3
Leptophlebiidae	Choroterpes		2	1	2	1	3	1
Leptophlebiidae	Leptophlebia		2	1	2	1	2	3
Leptophlebiidae	Paraleptophlebia		2	1	0	1	2	3
Oligoneuriidae			2	0	2	0	3	2
Oligoneuriidae	Homoeoneuria		2	0	2	0	3	1
Oligoneuriidae	Homoeoneuria	ammophila	2	0	2	0	3	1
Oligoneuriidae	Isonychia		2	0	2	0	3	2
Oligoneuriidae	Isonychia	rufa	2	0	2	0	3	2
Oligoneuriidae	Isonychia	sicca	2	0	2	0	4	2
Palingeniidae			3	0	2	0	3	4
Palingeniidae	Pentagenia		3	0	2	0	3	4
Palingeniidae	Pentagenia	vittigera	3	0	2	0	3	4
Polymitarcyidae			2	0	2	0	3	3
Polymitarcyidae	Ephoron		2	0	2	0	3	3
Polymitarcyidae	Ephoron	album	2	0	2	0	3	3
Polymitarcyidae	Tortopus		2	0	2	0	4	4
Polymitarcyidae	Tortopus	primus	2	0	2	0	4	4
Potamanthidae			2	1	2	1	3	3
Potamanthidae	Potamanthus		2	1	2	1	3	3
Potamanthidae	Potamanthus	myops	2	1	2	1	3	3
Potamanthidae	Potamanthus	rufus	2	1	2	1	3	3
Siphlonuridae			2	1	2	1	2	4
Siphlonuridae	Siphlonurus		2	1	2	1	2	4
Siphlonuridae	Siphlonurus	marshalli	2	1	2	1	2	4
Siphlonuridae	Siphlonurus	minnoi	2	1	1	1	2	4
Siphlonuridae	Siphlonurus	occidentalis	2	1	2	1	3	4
Tricorythidae			2	0	2	0	3	3
Tricorythidae	Tricorythodes		2	0	2	0	3	3
Tricorythidae	Tricorythodes	minutus	3	0	3	0	3	3
Tricorythidae	Tricorythodes	peridius	2	0	2	0	3	3

<u>HEMIPTERA</u>
(as of 10 December 1987)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Belostomatidae			2	4	2	3	3	3
Belostomatidae	Belostoma		3	4	3	3	3	3
Belostomatidae	Belostoma	bakeri	1	4	3	3	3	3
Belostomatidae	Belostoma	fluminea	4	4	3	3	3	3
Belostomatidae	Belostoma	lutarium	3	4	3	3	3	3
Belostomatidae	Lethocerus		2	4	1	3	3	3
Belostomatidae	Lethocerus	americana	2	4	1	3	3	3
Belostomatidae	Lethocerus	griseus	2	4	1	3	3	3
Belostomatidae	Lethocerus	uhleri	2	4	1	3	3	3
Corixidae			3	4	3	3	4	3
Corixidae	Callicorixa		4	4	3	3	4	3
Corixidae	Cenocorixa		4	4	3	3	4	3
Corixidae	Cenocorixa	utahensis	4	4	3	3	4	3
Corixidae	Corisella		4	4	3	3	4	3
Corixidae	Corisella	edulis	4	4	3	3	4	3
Corixidae	Corisella	tarsalis	3	4	3	3	4	3
Corixidae	Hesperocorixa		2	4	3	3	4	3
Corixidae	Hesperocorixa	laevigata	2	4	3	3	4	3
Corixidae	Hesperocorixa	nitida	2	4	3	3	4	3
Corixidae	Hesperocorixa	obliqua	4	4	3	3	4	3
Corixidae	Hesperocorixa	vulgaris	2	4	3	3	4	3
Corixidae	Palmacorixa		3	4	3	3	4	3
Corixidae	Palmacorixa	buenoi	3	4	3	3	4	3
Corixidae	Palmacorixa	gillettei	3	4	3	3	4	3
Corixidae	Palmacorixa	nana	3	4	3	3	4	3
Corixidae	Ramphocorixa		4	4	3	3	4	3
Corixidae	Ramphocorixa	acuminata	4	4	3	3	4	3
Corixidae	Sigara		3	4	4	3	5	2
Corixidae	Sigara	alternata	3	4	4	3	5	2
Corixidae	Sigara	grossolineata	3	4	4	3	5	2
Corixidae	Sigara	hubbelli	3	4	4	3	5	2
Corixidae	Sigara	modesta	3	4	5	3	5	2
Corixidae	Trichocorixa		3	4	3	3	4	3
Corixidae	Trichocorixa	calva	3	4	3	3	4	3
Corixidae	Trichocorixa	kanza	3	4	3	3	4	3
Corixidae	Trichocorixa	reticulata	3	4	3	3	4	3
Corixidae	Trichocorixa	sexcincta	3	4	3	3	4	3
Corixidae	Trichocorixa	verticalis	3	4	3	3	4	3
Gelastocoridae			4	3	3	3	3	4
Gelastocoridae	Gelastocoris		4	3	3	3	3	4
Gelastocoridae	Gelastocoris	oculatus	4	3	3	3	3	4
Gerridae			3	4	2	4	3	4
Gerridae	Gerris		3	5	3	5	4	4
Gerridae	Gerris	alacris	3	5	3	5	4	4
Gerridae	Gerris	argenticollis	2	5	3	5	4	4
Gerridae	Gerris	buenoi	1	5	3	5	4	4
Gerridae	Gerris	comatus	3	5	3	5	4	4

<u>Hemiptera (continued)</u>

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Gerridae	Gerris	insperatus	3	5	3	5	4	4
Gerridae	Gerris	marginatus	4	5	3	5	4	4
Gerridae	Gerris	nebularis	3	5	3	5	4	4
Gerridae	Gerris	remigis	3	5	5	5	4	4
Gerridae	Metrobates		3	4	2	3	3	4
Gerridae	Metrobates	hesperius	3	4	2	3	3	4
Gerridae	Metrobates	trux	3	4	2	3	3	4
Gerridae	Neogerris		3	4	2	3	3	4
Gerridae	Neogerris	hesione	3	4	2	3	3	4
Gerridae	Rheumatobates		3	4	2	3	3	4
Gerridae	Rheumatobates	hungerfordi	3	4	2	3	3	4
Gerridae	Rheumatobates	palosi	3	4	2	3	3	4
Gerridae	Rheumatobates	rileyi	3	4	2	3	3	4
Gerridae	Rheumatobates	trulliger	3	4	2	3	3	4
Gerridae	Trepobates		3	4	2	3	3	4
Gerridae	Trepobates	knighti	3	4	2	3	3	4
Gerridae	Trepobates	subnitidus	3	4	2	3	3	4
Hebridae			3	3	4	3	4	4
Hebridae	Hebrus		3	3	4	3	4	4
Hebridae	Hebrus	beameri	1	3	4	3	4	4
Hebridae	Hebrus	buenoi	3	3	4	3	4	4
Hebridae	Hebrus	burmeisteri	3	3	4	3	4	4
Hebridae	Hebrus	comatus	2	3	4	3	4	4
Hebridae	Hebrus	sobrinus	3	3	4	3	4	4
Hebridae	Merragata		4	4	3	3	4	4
Hebridae	Merragata	brunnea	3	4	3	3	4	4
Hebridae	Merragata	hebroides	4	4	3	3	4	4
Hydrometridae			4	5	4	4	3	4
Hydrometridae	Hydrometra		4	5	4	4	3	4
Hydrometridae	Hydrometra	australis	4	5	3	4	3	4
Hydrometridae	Hydrometra	hungerfordi	2	5	3	4	3	4
Hydrometridae	Hydrometra	martini	4	5	5	4	3	4
Mesoveliidae			3	5	4	5	3	4
Mesoveliidae	Mesovelia		3	5	4	5	3	4
Mesoveliidae	Mesovelia	cryptophila	2	5	3	5	3	4
Mesoveliidae	Mesovelia	douglasensis	1	5	3	5	3	4
Mesoveliidae	Mesovelia	mulsanti	4	5	4	5	3	4
Naucoridae			2	4	3	3	3	3
Naucoridae	Pelocoris		2	4	3	3	3	3
Naucoridae	Pelocoris	femoratus	2	4	3	3	3	3
Nepidae			2	4	3	3	3	4
Nepidae	Nepa		1	4	3	3	3	4
Nepidae	Nepa	apiculata	1	4	3	3	3	4
Nepidae	Ranatra		3	4	3	3	3	4
Nepidae	Ranatra	australis	2	4	3	3	3	4
Nepidae	Ranatra	fusca	4	4	3	3	3	4
Nepidae	Ranatra	kirkaldyi	2	4	3	3	3	4
Nepidae	Ranatra	nigra	3	4	3	3	3	4
Notonectidae			3	4	4	3	4	3

Hemiptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Notonectidae	Buenoa		3	4	3	3	4	3
Notonectidae	Buenoa	confusa	2	4	3	3	4	3
Notonectidae	Buenoa	margaritacea	4	4	3	3	4	3
Notonectidae	Buenoa	scimitra	2	4	3	3	4	3
Notonectidae	Notonecta		3	4	4	3	4	3
Notonectidae	Notonecta	indica	3	4	5	3	4	3
Notonectidae	Notonecta	irrorata	2	4	4	3	4	3
Notonectidae	Notonecta	undulata	4	4	4	3	4	3
Ochteridae			1	3	3	3	4	3
Ochteridae	Ochterus		1	3	3	3	4	3
Ochteridae	Ochterus	flaviclavus	1	3	3	3	4	3
Pleidae			3	3	2	3	3	3
Pleidae	Neoplea		3	3	2	3	3	3
Pleidae	Neoplea	striola	3	3	2	3	3	3
Saldidae			3	5	4	5	4	4
Saldidae	Micracanthia		4	5	4	5	4	4
Saldidae	Micracanthia	floridana	2	5	4	5	4	4
Saldidae	Micracanthia	humilis	4	5	4	5	4	4
Saldidae	Pentacora		3	5	4	5	4	4
Saldidae	Pentacora	ligata	3	5	4	5	4	4
Saldidae	Pentacora	signoreti	3	5	4	5	4	4
Saldidae	Salda		4	5	4	5	4	4
Saldidae	Salda	lugubris	4	5	4	5	4	4
Saldidae	Salda	provancheri	1	5	4	5	4	4
Saldidae	Saldoida		2	5	4	5	4	4
Saldidae	Saldoida	slossonae	2	5	4	5	4	4
Saldidae	Saldula		3	5	4	5	4	4
Saldidae	Saldula	comatula	3	5	4	5	4	4
Saldidae	Saldula	confluenta	3	5	4	5	4	4
Saldidae	Saldula	orbiculata	1	5	4	5	4	4
Saldidae	Saldula	pallipes	5	5	4	5	4	4
Saldidae	Saldula	pexa	2	5	4	5	4	4
Saldidae	Saldula	saltatoria	3	5	4	5	4	4
Saldidae	Saldula	severini	1	5	4	5	4	4
Veliidae			3	5	2	4	4	4
Veliidae	Microvelia		3	5	2	4	4	4
Veliidae	Microvelia	americana	4	5	2	4	4	4
Veliidae	Microvelia	cerifera	1	5	2	4	4	4
Veliidae	Microvelia	fontinalis	1	5	2	4	4	4
Veliidae	Microvelia	gerhardi	2	5	2	4	4	4
Veliidae	Microvelia	hinei	3	5	2	4	4	4
Veliidae	Microvelia	paludicola	2	5	2	4	4	4
Veliidae	Microvelia	pulchella	3	5	2	4	4	4
Veliidae	Paravelia		2	5	2	4	4	4
Veliidae	Paravelia	stagnalis	2	5	2	4	4	4
Veliidae	Rhagovelia		3	5	1	4	3	5
Veliidae	Rhagovelia	knighti	3	5	1	4	3	5
Veliidae	Rhagovelia	oriander	3	5	1	4	3	5
Veliidae	Rhagovelia	rivale	3	5	1	4	3	5

<u>LEPIDOPTERA</u>
(as of 10 December 1987)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Pyralidae			2	2	1	2	2	3
Pyralidae	Parapoynx		2	2	1	2	2	3
Pyralidae	Parapoynx	allionealis	2	2	1	2	2	3
Pyralidae	Petrophila		1	2	1	2	2	3
Pyralidae	Petrophila	bifascalis	2	2	1	2	2	3
Pyralidae	Petrophila	hodgesi	0	2	1	2	2	3
Pyralidae	Synclita		2	3	1	3	2	3
Pyralidae	Synclita	obliteralis	2	3	1	3	2	3

<u>MEGALOPTERA</u>
(as of 10 December 1987)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Corydalidae			2	2	3	2	3	3
Corydalidae	Chauliodes		2	2	2	2	3	4
Corydalidae	Chauliodes	pectinicornis	2	2	2	2	3	4
Corydalidae	Chauliodes	rastricornis	2	2	2	2	3	4
Corydalidae	Corydalus		2	2	5	2	3	3
Corydalidae	Corydalus	cornutus	2	2	5	2	3	3
Corydalidae	Nigronia		1	2	1	2	1	2
Corydalidae	Nigronia	serricornis	1	2	1	2	1	2
Sialidae			3	2	5	2	4	4
Sialidae	Sialis		3	2	5	2	4	4
Sialidae	Sialis	infumata	3	2	5	2	4	4
Sialidae	Sialis	itasca	3	2	5	2	4	4
Sialidae	Sialis	mohri	3	2	5	2	4	4
Sialidae	Sialis	vagans	3	2	5	2	4	4
Sialidae	Sialis	velata	3	2	5	2	4	4

<u>NEUROPTERA</u>
(as of 10 December 1987)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Sisyridae			2	2	1	2	3	3
Sisyridae	Climacia		2	2	1	2	3	3
Sisyridae	Climacia	areolaris	2	2	1	2	3	3
Sisyridae	Sisyra		1	2	1	2	3	3
Sisyridae	Sisyra	vicaria	1	2	1	2	3	3

ODONATA
(as of 10 December 1987)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Aeshnidae			2	4	1	3	3	4
Aeshnidae	Aeshna		3	4	1	3	3	4
Aeshnidae	Aeshna	constricta	3	4	1	3	3	4
Aeshnidae	Aeshna	multicolor	3	4	1	3	3	4
Aeshnidae	Aeshna	umbrosa	3	4	1	3	3	4
Aeshnidae	Anax		3	4	1	3	3	4
Aeshnidae	Anax	junius	3	4	1	3	3	4
Aeshnidae	Anax	longipes	2	4	1	3	3	4
Aeshnidae	Basiaeschna		2	4	1	3	3	2
Aeshnidae	Basiaeschna	janata	2	4	1	3	3	2
Aeshnidae	Boyeria		1	4	1	3	3	4
Aeshnidae	Boyeria	vinosa	1	4	1	3	3	4
Aeshnidae	Epiaeschna		1	4	1	3	3	3
Aeshnidae	Epiaeschna	heros	1	4	1	3	3	3
Aeshnidae	Nasiaeschna		1	4	2	3	3	3
Aeshnidae	Nasiaeschna	pentacantha	1	4	2	3	3	3
Calopterygidae			2	2	1	2	3	3
Calopterygidae	Calopteryx		2	2	2	2	3	3
Calopterygidae	Calopteryx	maculata	2	2	2	2	3	3
Calopterygidae	Hetaerina		3	2	1	2	3	3
Calopterygidae	Hetaerina	americana	3	2	1	2	4	3
Calopterygidae	Hetaerina	tita	2	2	1	2	2	2
Coenagrionidae			3	4	1	3	3	3
Coenagrionidae	Amphiagrion		1	4	1	3	4	2
Coenagrionidae	Argia		2	4	2	3	3	3
Coenagrionidae	Argia	alberta	1	4	1	3	3	3
Coenagrionidae	Argia	apicalis	3	4	2	3	3	3
Coenagrionidae	Argia	bipunctulata	1	4	1	3	3	3
Coenagrionidae	Argia	fumipennis	3	4	4	3	3	3
Coenagrionidae	Argia	moesta	3	4	3	3	3	4
Coenagrionidae	Argia	nahuana	2	4	1	3	3	3
Coenagrionidae	Argia	plana	2	4	3	3	2	2
Coenagrionidae	Argia	sedula	2	4	1	3	3	3
Coenagrionidae	Argia	tibialis	2	4	1	3	3	3
Coenagrionidae	Argia	translata	1	4	1	3	3	3
Coenagrionidae	Enallagma		3	5	1	3	3	3
Coenagrionidae	Enallagma	antennatum	3	5	2	3	3	3
Coenagrionidae	Enallagma	asperum	3	5	1	3	3	3
Coenagrionidae	Enallagma	basidens	3	5	2	3	3	3
Coenagrionidae	Enallagma	carunculatum	3	5	1	3	3	3
Coenagrionidae	Enallagma	civile	4	5	1	3	3	3
Coenagrionidae	Enallagma	divagans	3	5	1	3	3	3
Coenagrionidae	Enallagma	exulans	4	5	2	3	3	3
Coenagrionidae	Enallagma	geminatum	3	5	1	3	3	3
Coenagrionidae	Enallagma	praevarum	3	5	1	3	3	3
Coenagrionidae	Enallagma	signatum	3	5	2	3	3	3
Coenagrionidae	Enallagma	traviatum	3	5	1	3	3	3

<u>Odonata (continued)</u>

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Coenagrionidae	Enallagma	vesperum	3	5	4	3	3	3
Coenagrionidae	Ischnura		3	4	1	3	3	3
Coenagrionidae	Ischnura	barberi	1	4	1	3	5	3
Coenagrionidae	Ischnura	damula	2	4	1	3	3	3
Coenagrionidae	Ischnura	demorsa	2	4	1	3	3	3
Coenagrionidae	Ischnura	denticollis	3	4	1	3	3	3
Coenagrionidae	Ischnura	hastata	3	4	1	3	3	3
Coenagrionidae	Ischnura	perparva	3	4	1	3	3	3
Coenagrionidae	Ischnura	posita	3	4	1	3	3	3
Coenagrionidae	Ischnura	verticalis	4	4	1	3	3	4
Cordulegastridae			1	2	1	3	2	4
Cordulegastridae	Cordulegaster		1	2	1	3	2	4
Cordulegastridae	Cordulegaster	obliqua	1	2	1	3	2	4
Corduliidae			2	3	1	3	3	3
Corduliidae	Epicordulia		2	3	1	3	3	4
Corduliidae	Epicordulia	princeps	2	3	1	3	3	4
Corduliidae	Neurocordulia		2	3	1	3	3	3
Corduliidae	Neurocordulia	molesta	2	3	1	3	3	3
Corduliidae	Neurocordulia	xanthosoma	1	3	1	3	3	3
Corduliidae	Somatochlora		1	3	1	3	3	3
Corduliidae	Somatochlora	linearis	1	3	1	3	3	3
Corduliidae	Somatochlora	ozarkensis	1	3	1	3	3	3
Corduliidae	Somatochlora	tenebrosa	1	3	1	3	3	3
Corduliidae	Tetragoneuria		2	3	1	3	3	4
Corduliidae	Tetragoneuria	cynosura	3	3	1	3	3	4
Corduliidae	Tetragoneuria	williamsoni	2	3	1	3	3	4
Gomphidae			3	2	1	3	3	4
Gomphidae	Arigomphus		2	2	1	3	3	5
Gomphidae	Arigomphus	lentulus	2	2	1	3	3	5
Gomphidae	Arigomphus	submedianus	2	2	1	3	3	5
Gomphidae	Dromogomphus		3	2	1	3	3	4
Gomphidae	Dromogomphus	spinosus	3	2	1	3	3	4
Gomphidae	Dromogomphus	spoliatus	3	2	1	3	3	4
Gomphidae	Erpetogomphus		2	2	1	3	3	4
Gomphidae	Erpetogomphus	designatus	2	2	1	3	3	4
Gomphidae	Gomphus		2	2	1	3	3	5
Gomphidae	Gomphus	externus	3	2	1	3	3	5
Gomphidae	Gomphus	graslinellus	3	2	1	3	3	5
Gomphidae	Gomphus	militaris	3	2	1	3	3	5
Gomphidae	Gomphus	ozarkensis	2	2	1	3	3	5
Gomphidae	Gomphus	vastus	2	2	1	3	3	5
Gomphidae	Hagenius		1	2	1	3	3	3
Gomphidae	Hagenius	brevistylus	1	2	1	3	3	3
Gomphidae	Ophiogomphus		1	2	1	3	3	3
Gomphidae	Ophiogomphus	rupinsulenisis	1	2	1	3	3	3
Gomphidae	Ophiogomphus	severus	2	2	1	3	3	3
Gomphidae	Progomphus		2	2	1	3	3	3
Gomphidae	Progomphus	obscurus	2	2	1	3	3	3
Gomphidae	Stylogomphus		0	2	1	3	3	3

Odonata (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Gomphidae	Stylogomphus	albistylus	0	2	1	3	3	3
Gomphidae	Stylurus		2	2	1	3	3	3
Gomphidae	Stylurus	amnicola	3	2	1	3	3	3
Gomphidae	Stylurus	intricatus	2	2	1	3	3	3
Gomphidae	Stylurus	plagiatus	2	2	1	3	3	5
Lestidae			3	5	1	3	3	3
Lestidae	Archilestes		3	5	1	3	3	4
Lestidae	Archilestes	grandis	3	5	1	3	3	4
Lestidae	Lestes		3	5	1	3	3	3
Lestidae	Lestes	disjunctus	3	5	1	3	3	3
Lestidae	Lestes	rectangularis	3	5	1	3	3	3
Lestidae	Lestes	unguiculatus	3	5	1	3	3	3
Libellulidae			3	3	2	2	3	4
Libellulidae	Celithemis		1	5	2	5	3	3
Libellulidae	Celithemis	elisa	1	5	2	5	3	3
Libellulidae	Celithemis	eponina	1	5	2	5	3	3
Libellulidae	Celithemis	fasciata	1	5	2	5	3	3
Libellulidae	Dythemis		1	3	2	3	3	3
Libellulidae	Dythemis	fugax	1	3	2	3	3	3
Libellulidae	Erythemis		3	3	3	2	3	5
Libellulidae	Erythemis	simplicicollis	3	3	3	2	3	5
Libellulidae	Leucorrhinia		3	5	2	5	3	4
Libellulidae	Leucorrhinia	intacta	3	5	2	5	3	4
Libellulidae	Libellula		3	3	2	2	3	4
Libellulidae	Libellula	comanche	3	3	2	2	3	4
Libellulidae	Libellula	composita	2	3	2	2	3	4
Libellulidae	Libellula	cyanea	3	3	2	2	3	4
Libellulidae	Libellula	deplanata	3	3	2	2	3	4
Libellulidae	Libellula	flavida	2	3	2	2	3	4
Libellulidae	Libellula	incesta	3	3	2	2	3	4
Libellulidae	Libellula	luctuosa	4	3	2	2	3	4
Libellulidae	Libellula	pulchella	4	3	2	2	3	4
Libellulidae	Libellula	saturata	3	3	2	2	3	4
Libellulidae	Libellula	semifasciata	2	3	2	2	3	4
Libellulidae	Libellula	vibrans	2	3	2	2	3	4
Libellulidae	Orthemis		2	3	2	3	3	3
Libellulidae	Orthemis	ferruginea	2	3	2	3	3	3
Libellulidae	Pachydiplax		4	3	2	2	3	4
Libellulidae	Pachydiplax	longipennis	4	3	2	2	3	4
Libellulidae	Pantala		3	3	2	2	3	4
Libellulidae	Pantala	flavescens	3	3	2	2	3	4
Libellulidae	Pantala	hymenaea	3	3	2	2	3	4
Libellulidae	Perithemis		3	3	2	2	3	4
Libellulidae	Perithemis	tenera	3	3	2	2	3	4
Libellulidae	Plathemis		3	3	4	3	3	3
Libellulidae	Plathemis	lydia	4	3	5	3	3	3
Libellulidae	Plathemis	subornata	2	3	2	3	3	3
Libellulidae	Sympetrum		3	3	2	2	3	3
Libellulidae	Sympetrum	ambiguum	3	3	2	2	3	3

<u>Odonata (continued)</u>

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Libellulidae	Sympetrum	corruptum	3	3	2	2	3	3
Libellulidae	Sympetrum	costiferum	3	3	2	2	3	3
Libellulidae	Sympetrum	internum	3	3	2	2	3	3
Libellulidae	Sympetrum	obtrusum	3	3	2	2	3	3
Libellulidae	Sympetrum	occidentale	3	3	2	2	3	3
Libellulidae	Sympetrum	rubicundulum	3	3	2	2	3	3
Libellulidae	Sympetrum	vicinum	4	3	2	2	3	3
Libellulidae	Tramea		3	3	2	2	3	4
Libellulidae	Tramea	lacerata	3	3	3	2	3	4
Libellulidae	Tramea	onusta	3	3	2	2	3	4
Macromiidae			2	3	1	2	2	4
Macromiidae	Didymops		2	3	1	2	2	4
Macromiidae	Didymops	transversa	2	3	1	2	2	4
Macromiidae	Macromia		3	3	1	2	2	4
Macromiidae	Macromia	georgina	3	3	1	2	2	4
Macromiidae	Macromia	illinoiensis	3	3	2	2	2	4
Macromiidae	Macromia	pacifica	0	3	1	2	2	4
Macromiidae	Macromia	taeniolata	2	3	1	2	2	4
Petaluridae			1	3	1	2	3	3
Petaluridae	Tachopteryx		1	3	1	2	3	3
Petaluridae	Tachopteryx	thoreyi	1	3	1	2	3	3

PLECOPTERA

(as of 10 December 1987)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Capniidae			1	1	1	0	1	1
Capniidae	Allocapnia		1	1	1	0	1	1
Capniidae	Allocapnia	granulata	1	1	1	0	1	2
Capniidae	Allocapnia	mohri	0	1	1	0	1	0
Capniidae	Allocapnia	rickeri	1	1	1	0	1	1
Capniidae	Allocapnia	vivipara	2	1	1	0	1	2
Capniidae	Mesocapnia		0	2	1	0	1	1
Capniidae	Mesocapnia	frisoni	0	2	1	0	1	1
Capniidae	Paracapnia		1	1	1	0	1	1
Capniidae	Paracapnia	angulata	1	1	1	0	1	1
Chloroperlidae			0	1	0	0	0	1
Chloroperlidae	Alloperla		0	1	0	0	0	1
Chloroperlidae	Alloperla	hamata	0	1	0	0	0	1
Chloroperlidae	Haploperla		0	1	0	0	0	1
Chloroperlidae	Haploperla	brevis	0	1	0	0	0	1
Leuctridae			0	2	3	1	1	1
Leuctridae	Leuctra		0	2	2	1	1	1
Leuctridae	Leuctra	tenuis	0	2	2	1	1	1
Leuctridae	Zealeuctra		0	2	5	1	1	1
Leuctridae	Zealeuctra	claasseni	0	2	5	1	1	1
Leuctridae	Zealeuctra	narfi	0	2	5	1	1	1
Nemouridae			0	2	3	1	2	2
Nemouridae	Amphinemura		0	2	3	1	2	2
Nemouridae	Amphinemura	delosa	0	2	3	1	2	2
Nemouridae	Amphinemura	varshava	0	2	3	1	2	2
Perlidae			1	2	1	1	2	3
Perlidae	Acroneuria		0	1	1	0	2	3
Perlidae	Acroneuria	abnormis	0	1	1	0	2	4
Perlidae	Acroneuria	evoluta	0	1	1	0	1	2
Perlidae	Acroneuria	mela	1	1	1	0	2	4
Perlidae	Acroneuria	perplexa	0	1	1	0	2	2
Perlidae	Agnetina		0	1	1	0	3	2
Perlidae	Agnetina	flavescens	0	1	1	0	3	2
Perlidae	Attaneuria		1	2	1	1	3	2
Perlidae	Attaneuria	ruralis	1	2	1	1	3	2
Perlidae	Neoperla		1	2	2	1	3	2
Perlidae	Neoperla	catharae	1	2	2	1	3	2
Perlidae	Neoperla	choctaw	1	2	2	1	3	2
Perlidae	Neoperla	clymene	1	2	2	1	3	2
Perlidae	Neoperla	harpi	1	2	2	1	3	2
Perlidae	Neoperla	robisoni	1	2	2	1	3	2
Perlidae	Paragnetina		1	2	1	1	3	2
Perlidae	Paragnetina	kansensis	1	2	1	1	3	2
Perlidae	Perlesta		2	2	2	1	3	3
Perlidae	Perlesta	placida	2	2	2	1	3	3
Perlidae	Perlinella		0	2	1	1	2	2
Perlidae	Perlinella	drymo	0	2	1	1	2	2

Plecoptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Perlidae	Perlinella	ephyre	0	2	1	1	2	2
Perlodidae			1	2	1	1	3	2
Perlodidae	Clioperla		1	2	1	1	3	2
Perlodidae	Clioperla	clio	1	2	1	1	3	2
Perlodidae	Helopicus		1	2	1	1	3	2
Perlodidae	Helopicus	nalatus	1	2	1	1	3	2
Perlodidae	Hydroperla		1	2	1	1	3	2
Perlodidae	Hydroperla	crosbyi	2	2	2	1	2	2
Perlodidae	Hydroperla	fugitans	1	2	1	1	3	2
Perlodidae	Isoperla		1	2	1	1	3	2
Perlodidae	Isoperla	bilineata	1	2	1	1	3	2
Perlodidae	Isoperla	marlynia	1	2	1	1	3	2
Perlodidae	Isoperla	mohri	0	2	1	1	2	1
Perlodidae	Isoperla	namata	0	2	1	1	3	1
Perlodidae	Isoperla	ouachita	0	2	1	1	3	1
Perlodidae	Isoperla	quinquepunctata	1	2	1	1	3	2
Pteronarcyidae			1	3	3	2	3	3
Pteronarcyidae	Pteronarcys		1	3	3	2	3	3
Pteronarcyidae	Pteronarcys	pictetti	1	3	3	2	3	3
Taeniopterygidae			1	3	1	2	1	2
Taeniopterygidae	Strophopteryx		1	3	1	2	1	1
Taeniopterygidae	Strophopteryx	fasciata	1	3	1	2	1	1
Taeniopterygidae	Taeniopteryx		1	3	1	2	2	3
Taeniopterygidae	Taeniopteryx	burksi	2	3	1	2	2	3
Taeniopterygidae	Taeniopteryx	metequi	1	3	1	2	2	2

<u>TRICHOPTERA</u>
(as of 10 December 1987)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Brachycentridae			1	3	2	2	3	4
Brachycentridae	Brachycentrus		1	3	2	2	3	4
Brachycentridae	Brachycentrus	numerosus	1	3	2	2	3	4
Glossosomatidae			1	2	0	2	1	3
Glossosomatidae	Agapetus		1	2	0	2	1	3
Glossosomatidae	Agapetus	illini	1	2	0	2	1	3
Glossosomatidae	Glossosoma		1	2	0	2	1	3
Helicopsychidae			2	2	1	2	3	1
Helicopsychidae	Helicopsyche		2	2	1	2	3	1
Helicopsychidae	Helicopsyche	borealis	2	2	1	2	3	1
Helicopsychidae	Helicopsyche	piora	2	2	1	2	3	1
Hydropsychidae			3	3	2	2	3	3
Hydropsychidae	Cheumatopsyche		3	3	3	2	3	3
Hydropsychidae	Cheumatopsyche	aphanta	2	3	3	2	3	3
Hydropsychidae	Cheumatopsyche	campyla	4	3	4	2	4	4
Hydropsychidae	Cheumatopsyche	gracilis	1	3	3	2	1	3
Hydropsychidae	Cheumatopsyche	lasia	4	3	3	2	4	3
Hydropsychidae	Cheumatopsyche	miniscula	3	3	3	2	1	3
Hydropsychidae	Cheumatopsyche	oxa	2	3	3	2	3	3
Hydropsychidae	Cheumatopsyche	pasella	2	3	3	2	3	3
Hydropsychidae	Cheumatopsyche	pettiti	3	3	4	2	3	4
Hydropsychidae	Cheumatopsyche	rossi	2	3	3	2	4	3
Hydropsychidae	Diplectrona		0	3	0	2	1	1
Hydropsychidae	Diplectrona	modesta	0	3	0	2	1	1
Hydropsychidae	Hydropsyche		3	3	1	1	3	3
Hydropsychidae	Hydropsyche	arinale	2	3	1	1	2	3
Hydropsychidae	Hydropsyche	betteni	3	3	3	1	3	3
Hydropsychidae	Hydropsyche	bidens	3	3	1	1	3	3
Hydropsychidae	Hydropsyche	incommoda	2	3	1	1	2	3
Hydropsychidae	Hydropsyche	orris	3	3	2	1	3	3
Hydropsychidae	Hydropsyche	scalaris	3	3	1	1	2	3
Hydropsychidae	Hydropsyche	simulans	3	3	2	1	3	3
Hydropsychidae	Hydropsyche	valanis	2	3	1	1	3	3
Hydropsychidae	Potamyia		2	3	2	2	2	3
Hydropsychidae	Potamyia	flava	2	3	2	2	2	3
Hydropsychidae	Symphitopsyche		3	3	1	2	2	3
Hydropsychidae	Symphitopsyche	morosa	3	3	1	2	2	3
Hydropsychidae	Symphitopsyche	sparna	2	3	1	2	2	3
Hydroptilidae			3	2	2	1	2	3
Hydroptilidae	Hydroptila		3	2	3	1	3	3
Hydroptilidae	Hydroptila	ajax	2	2	3	1	3	3
Hydroptilidae	Hydroptila	angusta	3	2	3	1	2	3
Hydroptilidae	Hydroptila	armata	3	2	3	1	3	3
Hydroptilidae	Hydroptila	consimilis	2	2	3	1	3	3
Hydroptilidae	Hydroptila	grandiosa	3	2	3	1	1	3
Hydroptilidae	Hydroptila	pecos	3	2	3	1	3	3
Hydroptilidae	Hydroptila	perdita	3	2	2	1	1	3

Trichoptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Hydroptilidae	Hydroptila	rono	3	2	3	1	3	3
Hydroptilidae	Hydroptila	scolops	3	2	3	1	2	3
Hydroptilidae	Hydroptila	waubesiana	3	2	3	1	3	3
Hydroptilidae	Ithytrichia		2	2	1	1	2	2
Hydroptilidae	Ithytrichia	clavata	2	2	1	1	2	2
Hydroptilidae	Leucotrichia		3	2	1	1	2	2
Hydroptilidae	Leucotrichia	pictipes	3	2	1	1	2	2
Hydroptilidae	Mayatrichia		1	2	1	1	3	2
Hydroptilidae	Mayatrichia	ayama	1	2	1	1	3	2
Hydroptilidae	Neotrichia		3	2	1	1	2	2
Hydroptilidae	Neotrichia	falca	3	2	1	1	1	2
Hydroptilidae	Neotrichia	minutisimella	3	2	1	1	2	2
Hydroptilidae	Neotrichia	okopa	3	2	1	1	2	2
Hydroptilidae	Neotrichia	vibrans	3	2	1	1	1	2
Hydroptilidae	Ochrotrichia		3	2	2	1	2	4
Hydroptilidae	Ochrotrichia	anisca	3	2	2	1	1	4
Hydroptilidae	Ochrotrichia	tarsalis	3	2	2	1	2	4
Hydroptilidae	Orthotrichia		3	2	2	1	2	4
Hydroptilidae	Orthotrichia	aegerfasciella	3	2	2	1	2	4
Hydroptilidae	Orthotrichia	cristata	3	2	1	1	2	4
Hydroptilidae	Oxyethira		2	3	1	2	3	1
Hydroptilidae	Oxyethira	dualis	1	3	1	2	3	1
Hydroptilidae	Oxyethira	pallida	3	3	2	2	2	1
Hydroptilidae	Oxyethira	zeronia	2	3	1	2	3	1
Hydroptilidae	Stactobiella		2	3	1	2	0	1
Hydroptilidae	Stactobiella	delira	2	3	1	2	0	1
Hydroptilidae	Stactobiella	palmata	2	3	1	2	0	1
Leptoceridae			2	3	1	2	2	3
Leptoceridae	Ceraclea		2	2	1	1	1	4
Leptoceridae	Ceraclea	ancylus	2	2	1	1	1	4
Leptoceridae	Ceraclea	cancellata	2	2	1	1	1	4
Leptoceridae	Ceraclea	flava	2	2	1	1	1	4
Leptoceridae	Ceraclea	maculata	3	2	2	1	2	4
Leptoceridae	Ceraclea	neffi	2	2	2	1	1	4
Leptoceridae	Ceraclea	nepha	2	2	2	1	1	4
Leptoceridae	Ceraclea	protonepha	1	2	1	1	1	4
Leptoceridae	Ceraclea	spongillovorax	2	2	1	1	2	4
Leptoceridae	Ceraclea	tarsipunctata	2	2	1	1	1	4
Leptoceridae	Ceraclea	transversa	2	2	2	1	1	4
Leptoceridae	Leptocerus		2	3	1	2	3	3
Leptoceridae	Leptocerus	americanus	2	3	1	2	3	3
Leptoceridae	Nectopsyche		3	3	2	2	2	3
Leptoceridae	Nectopsyche	albida	3	3	2	2	2	3
Leptoceridae	Nectopsyche	candida	3	3	2	2	3	3
Leptoceridae	Nectopsyche	diarina	3	3	2	2	3	3
Leptoceridae	Nectopsyche	exquisita	2	3	2	2	1	3
Leptoceridae	Nectopsyche	pavida	2	3	2	2	1	3
Leptoceridae	Nectopsyche	spiloma	2	3	2	2	2	3
Leptoceridae	Oecetis		2	3	1	2	3	3

Trichoptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Leptoceridae	Oecetis	avara	2	3	1	2	2	3
Leptoceridae	Oecetis	cinerascens	2	3	1	2	3	3
Leptoceridae	Oecetis	ditissa	2	3	1	2	3	3
Leptoceridae	Oecetis	eddlestoni	2	3	1	2	3	3
Leptoceridae	Oecetis	inconspicua	2	3	1	2	2	3
Leptoceridae	Oecetis	nocturna	2	3	1	2	2	3
Leptoceridae	Oecetis	persimilis	1	3	1	2	1	3
Leptoceridae	Triaenodes		2	3	1	2	4	3
Leptoceridae	Triaenodes	injusta	2	3	1	2	4	3
Leptoceridae	Triaenodes	tarda	2	3	1	2	4	3
Limnephilidae			2	3	1	2	3	2
Limnephilidae	Ironoquia		2	3	2	2	3	2
Limnephilidae	Ironoquia	punctatissima	2	3	2	2	3	2
Limnephilidae	Limnephilus		1	3	1	2	3	2
Limnephilidae	Limnephilus	diversus	1	3	1	2	3	2
Limnephilidae	Limnephilus	taloga	1	3	1	2	3	2
Limnephilidae	Pycnopsyche		2	2	1	2	3	3
Philopotamidae			1	2	1	1	2	4
Philopotamidae	Chimarra		2	2	1	1	2	4
Philopotamidae	Chimarra	feria	1	2	1	1	2	4
Philopotamidae	Chimarra	obscura	2	2	1	1	2	4
Philopotamidae	Wormaldia		0	2	0	1	0	0
Phryganeidae			2	3	2	2	4	3
Phryganeidae	Agrypnia		2	3	2	2	4	3
Phryganeidae	Agrypnia	vestita	2	3	2	2	4	3
Phryganeidae	Phryganea		2	3	3	2	4	3
Phryganeidae	Phryganea	sayi	2	3	3	2	4	3
Polycentropodidae			2	3	2	2	2	2
Polycentropodidae	Cernotina		2	3	1	2	2	3
Polycentropodidae	Cernotina	calcea	2	3	1	2	2	3
Polycentropodidae	Cernotina	spicata	2	3	1	2	2	3
Polycentropodidae	Cyrnellus		3	2	2	1	3	2
Polycentropodidae	Cyrnellus	fraternus	3	2	2	1	3	2
Polycentropodidae	Neureclipsis		2	3	1	2	2	3
Polycentropodidae	Neureclipsis	crepscularis	2	3	1	2	2	3
Polycentropodidae	Nyctiophylax		2	3	1	2	2	2
Polycentropodidae	Nyctiophylax	affinis	2	3	1	2	2	2
Polycentropodidae	Polycentropus		2	3	4	2	1	1
Polycentropodidae	Polycentropus	centralis	1	3	5	2	1	1
Polycentropodidae	Polycentropus	cinereus	2	3	2	2	2	1
Polycentropodidae	Polycentropus	crassicornis	2	3	5	2	1	1
Polycentropodidae	Polycentropus	nascotius	1	3	2	2	3	1
Psychomyiidae			1	2	1	1	0	1
Psychomyiidae	Psychomyia		1	2	1	1	0	1
Psychomyiidae	Psychomyia	flavida	1	2	1	1	0	1
Rhyacophilidae			0	3	1	2	2	2
Rhyacophilidae	Rhyacophila		0	3	1	2	2	2
Rhyacophilidae	Rhyacophila	lobifera	0	3	1	2	2	2
Sericostomatidae			0	3	1	2	2	1

Trichoptera (continued)

FAMILY	GENUS	SPECIES	NOD	AP	HM	POC	SA	SSS
Sericostomatidae	Gumaga		0	3	1	2	2	1
Sericostomatidae	Gumaga	griseola	0	3	1	2	2	1