Basin Visual Estimation Technique (BVET) and Representative Reach Approaches to Wadeable Stream Surveys: Methodological Limitations and Future Directions

ABSTRACT

Basin Visual Estimation Techniques (BVET) are used to estimate abundance for fish populations in small streams. With BVET, independent samples are drawn from natural habitat units in the stream rather than sampling "representative reaches." This sampling protocol provides an alternative to traditional reach-level surveys, which are criticized for their lack of accuracy in estimating abundance at larger scales. BVET methodologies have been adopted and used by numerous government agencies for monitoring stream biota. Many of the assumptions of BVET methods, however, cannot be met in streams where they are being implemented because of unsuitable conditions for BVET surveys. Lack of bed control structures, variability in flow regimes, and lack of consistency among observers create difficulties in assessing habitat using BVET methods. BVET methods also are used to assess assemblage structure in streams although that was not the application for which they were originally designed. Representative reach approaches also have problems, as they often do not accurately reflect conditions present throughout the stream. We review various studies in which BVET and representative reach methodologies were employed and make recommendations for their most appropriate application given a range of study objectives.

Lance R. Williams
Melvin L. Warren, Jr.
Susan B. Adams
Joseph L. Arvai
Christopher M. Taylor

Williams is an assistant professor in the School of Natural Resources, Ohio State University, Columbus. He can be reached at williams.2323@osu.edu. Warren and Adams are research biologists in the USDA Forest Service, Center for Bottomland Hardwoods Research, Southern Research Station, Oxford, MS. Arvai is an assistant professor, School of Natural Resources, Ohio State University. Taylor is an associate professor in the Department of Biological Sciences, Mississippi State University, Mississippi State, MS.

Introduction

In 1988, Hankin and Reeves presented an alternative to the long-standing "representative reach" approach for estimating fish abundances in wadeable streams. Their methodology has since been used by state and federal government agencies and other researchers to estimate abundances of individual species (Hankin and Reeves 1988; Toepfer et al. 2000), to assess biota-habitat relationships (Dolloff et al. 1997; Leftwich et al. 1997; Peterson and Rabeni 2001), to develop instream flow criteria (Kershner and Snider 1992), or to examine land-use impacts on aquatic biota (Clingenpeel and Cochran 1992; Ensign et al. 1997; Williams et al. 2002).

Hankin and Reeves (1988) argued that sampling fishes at a fixed number of reaches and then extrapolating up to the entire stream would lead to large errors in estimates of abundance. Instead, they proposed a more statistically rigorous methodology in which independent samples were drawn from natural habitat units in the stream. The study system where this method was developed is Cummins Creek, a small upland stream in west-central Oregon. Hankin and Reeves stratified habitat units by type (e.g., riffle, pool, run) and location (lower, mid, and upper reaches) within the stream and then visually estimated area for each of the habitat units. Within each habitat type-by-location combination, a systematic sample of n units was snorkeled to enumerate fish. Multiple-pass electrofishing and actual measurement of unit area were conducted in a subsample of these units to provide a "true" fish count for comparison with snorkeling data and a "true" areal measurement for comparison with visual estimates of unit areas. The method originally was applied to populations of coho salmon (*Oncorhynchus kisutch*) and steelhead trout (Oncorhynchus *mykiss*) in Cummins Creek. Using these methods, Hankin and Reeves (1988) produced detailed maps of the stream, and demonstrated that target fish abundances varied with the size and composition of habitat units. They concluded that extrapolating from only a few representative reaches would have produced misleading abundance estimates for the salmonids in their study system.

We provide a critical review of both Basin Visual Estimation Technique (BVET) and representative reach approaches to sampling wadeable stream systems. In particular we address some of the assumptions of these Iwo approaches and comment on their applicability for a range of study objectives in different types of stream systems.

Habitat Assessment with BVET

A basic tenet in stream ecology is that organisms respond to variation in the structure of the physical environment (Vannote et al. 1980; Minshall 1988). Because the physical environment varies greatly at the scale of reaches and channel units (Matthews et al. 1994; Taylor 1997; D'Angelo et al. 1997), these are appropriate scales to measure habitat in wadeable streams. Streams traditionally have been sampled at individual reaches, often in areas that may not he representative of the system (e.g., hridges). Also, the lengths of reaches have been defined differently, ranging from tens (Frissell et al. 1986) to hundreds of meters (Lyons 1992; Angermeier and Smogor 1995; Paller 1995). Thus, much of our current understanding of the ecology of streams is limited to patterns obtained from limited, reach-scale sampling (Fausch et al. 2002; Lowe 2002). Sampling streams at reaches that are assumed to he representative of the entire stream condition has fallen into disfavor in recent years. Advances in landscape ecology have shown that species' life histories and their responses to environmental perturbations operate at much larger scales (Schlosser 1991, 1995; Fausch et al. 2002; Lowe 2002).

In an attempt to understand and inventory habitat at the whole-stream scale, classification by channel geomorphic units (CGUs) has become an important management tool. Dividing streams into CGUs is used when measuring distribution, abundance, and assemblage structure of fishes and invertebrates (Frissell et al. 1986; Armitage et al. 1995; Raheni et al. 2002). Channel geomorphic unit characterization also is used extensively in biological monitoring programs by federal (e.g., EPA, USGS National Water Quality Assessment [NAWQA] program, and the USDA Forest Service) and state agencies. The ability to visually identify CGUs accurately is a cornerstone in Hankin and Reeves' (1988) methodology.

Most CGU methods classify streams into natural geomorphic units determined primarily by depth and flow and secondarily by substrate composition. Hawkins et al. (1993) devised a classification based on fast ("riffle") and slow ("pool") categories of current velocity. Riffles were further divided into "highly turbulent" and "low turbulence" categories, and pools were sub-divided into "channel scour" and "formed behind dams" groups (Figure 1). Overall, Hawkins et al. (1993) recognized 18 different CGUs in streams. Other classifications have been proposed (e.g., Bisson et al. 1982; McCain et al. 1990), hut most are similar to Hawkins et al.'s model.

A major criticism of these methods is the difficulty in applying the theoretical definitions of various hahitat units to streams in the field. A numher of authors expressed concern about the repeatability and precision of CGU classification (Platts et al. 1983; Dolloff et al. 1993; Roper and Scharnecchia 1995). In a review of CGU methods and their application, Poole et al. (1997) noted serious problems with observer bias. Using previously published data, they calculated that sub-classification of habitat types by field ohservers (the finest scale, Figure 1) was only 29% to 56% better than randomly classifying the habitat units. As the number of habitat types used to classify a stream decreases, consistency among observers tends to increase, but even when classification types are simplified (e.g., 3 primary types), observers are in agreement only 75% of the time (Roper and Scarnecchia 1995). Of course, these methods are not applicable to large river systems as classifying even riffle-pool-run sequences is usually untenable.

We see differences among observers as a substantial impediment to applying BVET methods, particularly in systems where assigning CGUs is highly subjective, especially prone to biased judgments, or simply inapplicable. In many of the Coastal Plain systems with which the authors are most familiar, huge sections of streams would he classified as "runs" when there are obviously a myriad of microhabitats embedded in this coarse category. For example, about 54% of 371 reaches (20 to 100 m long) in small, sand-bed channel streams in Mississippi national forests were classified over their entire lengths as runs (7.8 km of 13.2

FAST
("Riffle")

Highly Turbulent
1. Falls
2. Cascades
3. Chutes
4. Rapids
5. Riffles

Low Turbulence
1. Sheets
2. Runs

SLOW
("Pool")

Channel Scour Pools
1. Eddy
2. Trench
3. Mid-channel
4. Convergence
5. Lateral Scour
6. Plunge

Pools Behind Dams
1. Debris
2. Beaver
3. Landslide
4. Backwater
5. Abandoned Channel

Figure 1. Summary of Hawkins et al. (1993) channel geomorphic units (CGUs) classification scheme.

km total stream length; Warren unpublished data; Warren et al. 2002; Figure 2). However, the physical variability within runs was high as measured by cross-sectional transects (e.g., mean depths 1.3 to 54.9-cm, coefficient of variation [CV] 26 -125%; mean velocities 2 to 71 -cm/s, CV 12 -119%). The range of mean depths and velocities measured by transects within runs broadly overlapped values observed in reaches classified as pools or riffles (Warren unpublished data; Warren et al. 2002). Sand-bed streams often lack bed control structures (e.g., bedrock, boulders, large woody debris) necessary for fine-scale delineation of CGUs, and are composed predominantly of sand, silt, and gravel substrates that are very "fluid" and can shift considerably with variation in stream flow (Ross et al. 2001; Figure 3). Because of difficulties associated with assigning CGUs, we yuestion the utility of fine or even coarse-scale classifications in these kinds of habitats. In sand-bed Coastal Plain streams, most of the 18 categories (Figure 1) recognized by Hawkins et al. (1993) are not applicable.

Another practical consideration in applying BVET methods that has received little attention is defining the minimum size (e.g., length) of habitat units to be measured. In our review, we found only one paper (Clingenpeel and Cochran 1992) that identified a minimum size (10 m) for CGU classification, and we are aware of an unpublished "rule of thumb" that minimum units be "longer than wide" (MLW, personal observation). The questions then become how small is small enough for a given objective, and with finite resources, how much time and effort can be spent per unit length of a stream? If the minimum CGU definition is large, then conditions within a given unit could be highly variable. If the minimum unit is very small, then classification crews could spend hours in a short length of stream trying to visually estimate (or measure) numerous "micro"-CGUs. Moreover, the ideal minimum CGU size changes with stream size.

Further problems arise concerning criteria for streams selected for CGU application. Classification of CGUs is recommended for wadeable streams during base or late summer flow and is designed for natural, perennial streams free of human-caused impacts (Roper and Scarnecchia 1995; Arend 1999). These conditions can be difficult to find in the field, and limiting sampling to these types of streams and conditions would yield no information on how seasonal variability in discharge influences habitat classification. CGUs can change dramatically from peak flow in the winter or spring to base flow in the summer. Even if CGUs are classified only during summer base flows, our experience in eastern streams indicates CGUs undergo changes in a matter of weeks during summer dry down (Taylor and Warren 2001; Williams et al. 2003a). For example, Williams et al. (2003a) sampled 12 pools in a small Ouachita Mountain stream in Arkansas during summer 1999. Within 42 days, these pools went from a flowing system interconnected by riffles and runs to a series of isolated pools. By the end of this period, 8 of the 12 pools had dried completely with no interstitial flow. Conversely, substantial pools in sand-bed streams can be completely obliterated after several routine storm events (S. Adams, unpublished data). These changes in habitat structure will significantly alter use by stream biota, and thus, necessitate that minimal time elapses between habitat and fish sampling under a BVET approach.

Because CGUs are dependent on flow regime, some investigators have questioned their use to monitor human-caused impacts (Ralph et al. 1994; Roper and Scarnecchia 1995; Poole et al. 1997). Poole et al. (1997) concluded that changes in the frequency of occurrence (e.g., riffle-to-pool ratio) or relative area (e.g., pool size) of units are inconsistent and often insensitive measures of stream impact. Ralph et al. (1994) argued that measures of woody debris provided a better estimate of logging impacts than changes in habitat units. In streams with flow alteration or regulation caused by dams, diversions, or withdrawals, flow can be substantially modified over short periods of time, which can bias classification of CGUs and quantification of habitat. Because of the difficulties in accurately and precisely assigning CGUs, Roper and Scarnecchia (1995) argued that it might be more ecologically meaningful to conduct fewer, more rigorous studies in specific reaches than in applying a complex habitat classification scheme to a whole stream.

Assessment of Biota with BVET

Although BVET methods have been used extensively in the field to assess biota of wadeable streams, their application has not been without problems. While BVET methods may be effective in small, clear streams like Cummins Creek (Hankin and Reeves 1988), they become problematic when water clarity is low or when applied to assemblage-level studies (Williams et al. 2002). In southern U.S. Piedmont and Coastal Plain streams, water clarity often is not amenable to counting fish by snorkeling. Another difficulty with snorkel surveys is the high diversity and abundance of stream fishes found in much of the eastern United States as compared with the Pacific Northwest (Warren and Burr 1994; Matthews 1998; Warren et al. 2000). In many coldwater streams, snorkeling efficiency, for many salmonids at least, is temperature dependent (Hillman et al.1992; Thurow 1994), such that even in summer, morning water temperatures in high elevation streams may be too low for maximal efficiency.

In most BVET studies, three or fewer species are the focus of the snorkeling counts (e.g., Hankin and Reeves 1988; Ensign et al. 1997), and generally the targeted species are large (>10 cm, Thurow 1994). Snorkeling can be particularly unreliable for small fishes occupying shallow water, especially where substrate is large (Cunjak et al. 1988). Even small eastern streams can have local species richness exceeding 10–20 species (Matthews 1998), most of which are small-bodied (<10 cm) even as adults. The diversity and size of fishes makes using BVET for assemblage-level studies nearly impossible in many warmwater streams unless field crews consist of highly trained ichthyologists. Even then, benthic taxa, such as darters (*Etheostomu*, *Ammocryp ta*, *Percina*), madtom catfishes (*Noturus*), and sculpins (*Cottus*), which make up a large part of eastern stream fish assemblages, are small, cryptically colored, often nocturnal feeders, and, as a result, are difficult to see and accurately count via snorkeling. Water column dwelling species in eastern streams, such as *Notropis* and *Cyprinella*, can occur in mixed schools of hundreds of individuals, making accurate visual counts difficult for even an experienced diver.

Although they typically have lower species richness, streams in the western United States also can be difficult to assess using RVET methods. Western U.S. streams also can have one or more non-salmonid species (e.g., sculpin, dace, or minnow species) that are typically ignored during snorkel surveys or dropped from data analyses. Thurow (1994) suggested eliminating most age-0 salmonids from analyses because of difficulties with underwater identification. Finally, Hillman et al. (1992) found that coho and Chinook salmon (0. *tshawytscha*) in mixed groups of more than 40 fish were underestimated at least 50% of the time by snorkelers.

In larger streams, snorkeling becomes even more difficult, if not impossible, for assessing biota (e.g., more diversity, often poorer water clarity), and accurate electroshocking estimates cannot he made to verify the snorkeling estimates. Thompson (2003) found that BVET methods will yield poor results unless electroshocking removal estimates exceed 85% of the true numbers of fish within habitat units, and he recommended that correlations (r) between removal estimates and snorkel counts he at least 0.9. In all hut the smallest streams, these conditions would he extremely difficult, if not impossible, to meet.

The requirement of electroshocking to verify snorkeling estimates itself can he problematic. There are times when electroshocking cannot he used because of endangered species issues, inappropriate conductivities, or stream access problems. In many streams, electrofishing removal estimates are biased low (Peterson et al. 2004) and fish movement out of sample units can he relatively high,

Figure 2. USDA Forest Service personnel quantify habitat in a sand-bed Mississippi stream.

Figure 3. Typical sand-bed stream in Louisiana showing lack of bed-control structures and ubiquitousness of pool-run habitat.

further demonstrating the problem of using electroshocking to validate snorkeling estimates. Electroshocking also can be difficult in large streams or in areas with abundant undercut banks and woody debris (Figure 4). For some species-habitat combinations, snorkeling can be more efficient than electrofishing (reviewed in Thurow 1994; Ensign et al. 1995), rendering the latter useless as a verification tool. Furthermore, the relationship between snorkeling and electrofishing estimates may he confounded by hourly and daily fluctuations in water temperatures.

Some investigators have modified BVET methods in an attempt to more accurately assess hiota. Toepfer et al. (2000), for example, modified BVET methods to estimate abundance of the federally threatened leopard darter (*Percina pantherina*) in an Oklahoma stream, Big Eagle Creek. One difficulty they encountered was that Cummins Creek (from Hankin and Reeves 1988) comprised only 6% of the area of Big Eagle Creek. Species richness in Big Eagle Creek dwarfed that of Cummins Creek, making identification of even the target species problematic. Also, assigning channel geomorphic units in a stream of the size and complexity of Big Eagle Creek can be difficult as CGUs in the same class can differ considerably (Warren unpublished; Warren et al. 2002). Vadas and Orth (1998) argued that if there are problems with implementing a CGU classification scheme (or disagreement over particular habitat types like runs versus glides), microhabitat data should be quantified in some way. Modifications made by Toepfer et al. (2000) included sampling microhabitat within a selected group of CGUs using a transect design proposed by Simonson et al. (1994) for stream reaches. Cross sectional transects were established systematically along the thalweg within a reach (or CGU) and mean stream width (MSW) was used to establish length of sampling reach and number of transects. Simonson et al. (1994) recommended that reach length he approximately 35–40 MSW, and transect spacing should be every 2 or 3 MSW depending on the width of the stream (if width is greater than 5 m then transects should be spaced every 2 MSW; if stream width is less than 5 m then spacing should be every 3 MSW). Using the transect approach, Toepfer et al. (2000) were able to target specific CGUs that would contain suitable habitat for the leopard darter rather than trying to assess the entire stream.

Another difficulty with applying BVET methods is related to the inherent longitudinal distribution of stream habitat types and biota (Sheldon 1968; Vannote et al. 1980) that influence applicability of sampling methods. Channel geomorphic units in headwaters are less predictable, both physically and biotically, than in downstream reaches (Peterson and Rabeni 2001). In headwaters, CGUs are smaller and often subject to more severe natural disturbance regimes (e.g., variation in flow) than downstream CGUs. Thus, hcadwater faunas frequently consist of more generalist species (Peterson and Rabeni 2001).

Because generalists typically show weaker relationships to habitat complexity (Kolasa 1989), using BVET methods to relate headwater faunas to their habitat can he difficult. Based on data collected using BVET methods (Clingenpeel and Cochran 1992), Williams et al. (2002, 2003b) assessed impacts of timber harvesting and habitat conditions on the biota of small streams in the Ouachita Mountains, Arkansas. While these methods were effective for determining fish and macroinvertebrate-habitat relationships at the whole-stream scale, dividing individual streams into longitudinal segments for analysis was problematic. Clingenpeel and Cochran (1992) collected habitat data

Figure 4. Electroshocking for salmonids in Michigan's Upper Peninsula. Large amounts of woody debris can make electroshocking difficult, if not impossible, to accurataely estimate fish abundances.

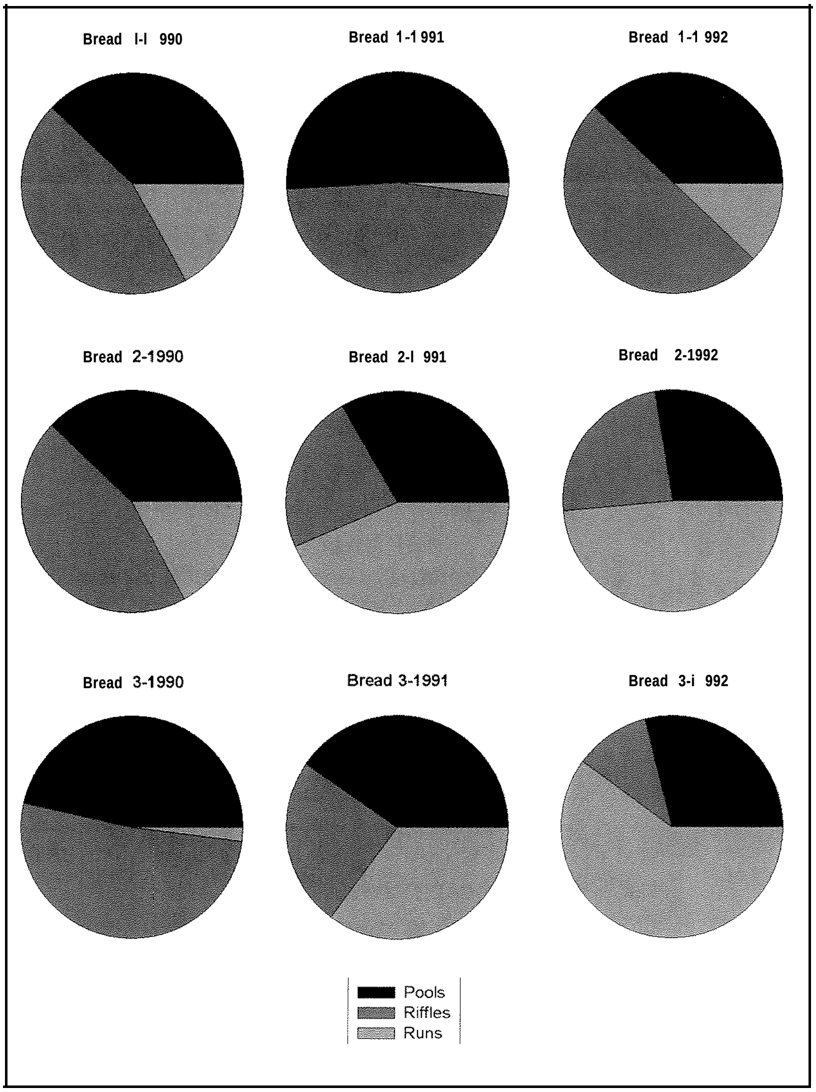
throughout 6 streams for 3 years (1990-1992) and sampled fishes and macroinvertebrates in 10% of each CGU type (e.g., if a stream had 300 mid-channel pools, then biota were sampled in 30). Although they attempted to stratify samples longitudinally, some stream sections were inadequately sampled. For example, rare hahitats in headwaters of these streams (e.g., riffles) were poorly represented. Although less common, headwater riffles are important hahitats for some species (e.g., redfin darter, *Etheostoma whipplei*, and the endemic Ouachita madtom, *Noturus lachneri*) in these Ouachita Mountain streams. Thus, for some objectives, a higher percentage or minimum numher of each rare habitat type needs to he sampled.

Williams et al. (2002) originally attempted to divide the same Ouachita streams into 3 to 4 equal-length segments for analysis, hut encountered problems with under-represented habitats and dramatic changes over time within the same stream sections. In their original surveys, Clingenpeel and Cochran (1992) recognized 24 types of CGUs. To reduce variahility, these were aggregated into 3 classes (riffles, pools, and runs). Because all streams were sampled during the same summer periods, we expected some consistency in the proportion of CGUs hy stream segment. For example, however, the most downstream segment of Bread Creek (Bread 3; Figure 5) had minimal run habitat in 1990, hut in 1992 runs were the dominant CGU. The obvious question is, “What is the source of the variability in CGUs over time: observer bias, flow differences in the summer, or hahitat change because of logging activities, etc.?” We found no evidence of logging impacts on biota in these streams, and current velocity did not change significantly from year-to-year (Williams et al. 2002; Williams et al. 2003b). Additionally, bedrock and large boulders dominate these streams; thus, structure of the streambed does not change significantly over several years. Therefore, we assume that inconsistencies in the proportion of CGUs over time are the result of observer bias. Because of limitations of the BVET methodology, Williams et al. (2002, 2003b) could conduct analyses only at the whole-stream scale. While there are obvious problems with scaling up to the whole stream from representative reaches, the reverse problems exist for BVET sampling—scaling down from the whole stream to individual reaches is not possible if an insufficient number of CGUs are sampled.

Representative Reach Approaches

Applying representative reach approaches to sampling habitat and biota also has been problematic. Ensign et al. (1997) found that representative reach methods did not work well for species with low abundances or clumped spatial distributions. Dolloff et al. (1997) also compared BVET methods to a representative reach approach in southern Appalachian streams and found BVET estimates were more accurate in identifying fish habitat. With the representative reach approach, average areas for habitat types, depth estimates, and measures of large woody debris tended to he smaller than the actual values in the stream (Dolloff et al. 1997). In addition, BVET methods were more likely to identify uncommon habitat units in a stream (e.g., cascades; Dolloff et al. 1997). Simonson (I 993), however, compared transect-based, representative reach sampling to BVET methods to assess fish habitat and found the reach approach superior because of the increase in precision. Using a hierarchical model in Oregon streams, D’Angelo et al. (1997) found fish distri-

Figure 5. Proportion of pools, riffles, and runs in three sections of Bread Creek, Arkansas during summer 1990-1992. Section 1 represents the headwaters and section 3 the most downstream reach; sections represent equal stream lengths.



butions were predicted accurately at the reach scale, but patterns for invertebrates and algae were evident only at the scale of CGUs.

Much of the disagreement over the effectiveness of reach-level sampling for quantifying habitat-hiota relationships is probably **related to how** reaches are defined. As a result, much effort recently has focused on the minimum length of a sampling reach to **he** representative of stream conditions. Most authors agree that the length of stream sampled must approach or exceed the length at **which** cumulative species richness becomes asymptotic (Lyons 1992; Angermeier and Smogor 1995; Paller 1995). Lyons (1992) recommended that stream widths be used to determine the length of a sampling reach, rather than using sequences of CGUs. Simonson et al. (1994) recommended that reach length **he** approximately 3 5-40 MSWs, and transects within each reach should **he spaced** according to stream width. Paller (1995), using data from southeastern U.S. streams, also determined that stream width should **he used** to estimate reach length, and recommended a reach length of 235 to 555 -m (35-l 58 MSWs). In smaller streams, higher ratios of reach length-to-width were necessary to accurately measure species richness. Angermeier **and** Smogor **(1995) found** that 90% of the species present **in a stream could be found hy** sampling a stream length of 22-67 MSWs.

The major problem in determining whether a stream reach is representative is that it requires detailed habitat assessment prior to determining the area to sample. Also, determining when species richness **becomes** asymptotic may require an **enor**mous amount of sampling. Some small streams may require sampling in entirety for the species accumulation curve to become asymptotic, or to sample a sufficient number of stream widths (e.g., 158 according to Paller 1995). Also, species accumulation curves are biased because they cannot account for incomplete detectability (Cam et al. 2002). Because of these limitations, it is probably unrealistic to ever expect reaches to **he completely** representative of stream conditions. A better, more statistically defensible approach would **he** a stratified random reach approach, where stream reaches are stratified **hy** some criteria (e.g., CGU types or stream segments [sensu Frissell et al. 1986]) or an adaptive sampling design (Smith et al. 2000). Still, considerable lengths of streams may need to **he sampled** for a representative reach approach to accurately reflect conditions in **the system.**

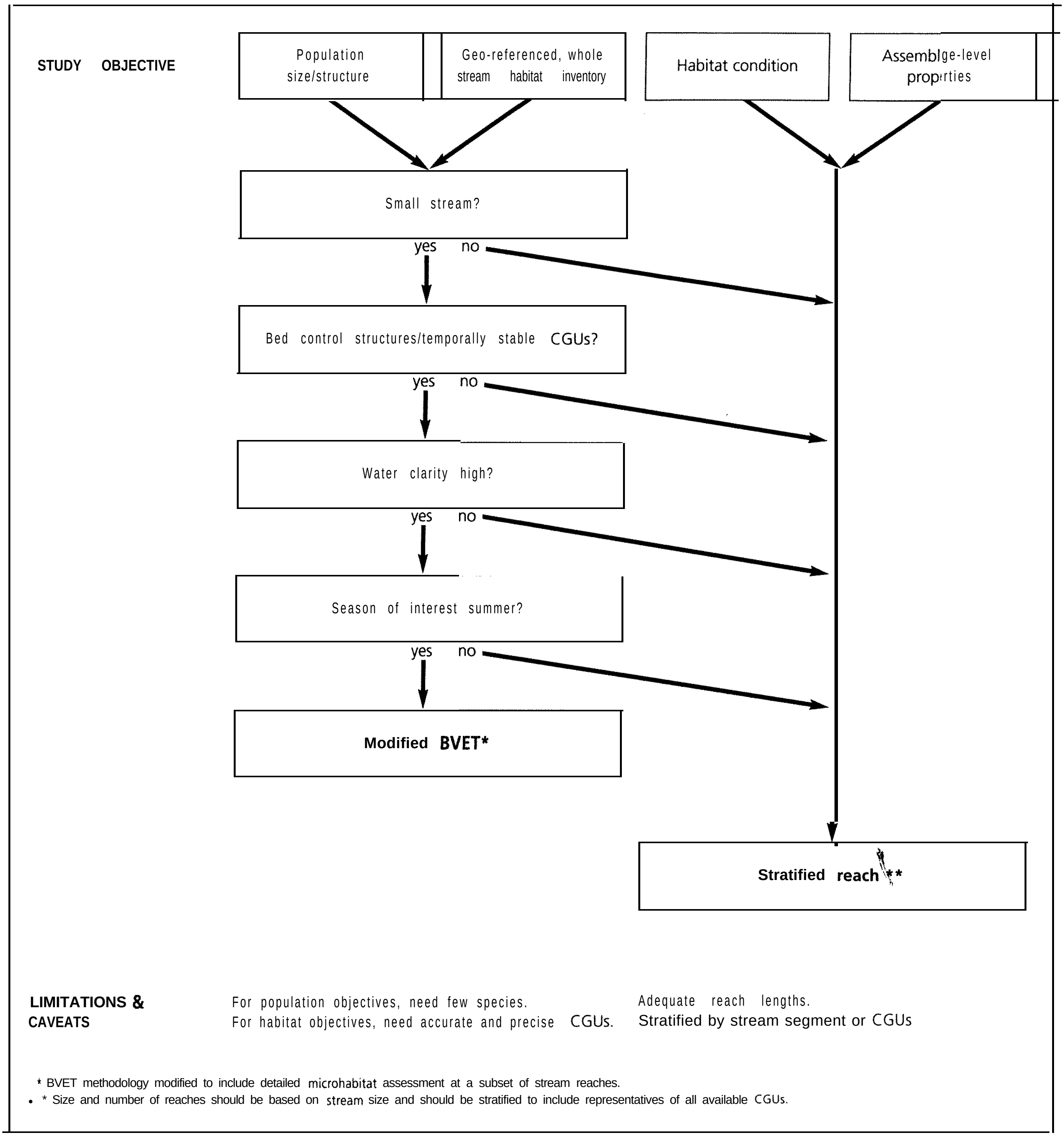
Recommendations and Conclusions

Given the inherent variability of stream systems, any sampling methodology will have problems. The key is to design and use methods that maximize accuracy and precision of the data for the specific objective while minimizing the cost and effort. The BVET and reach approaches have limitations, but both methodologies can be used effectively to assess and monitor stream hiota. What is required of investigators is a clear set of objectives, knowledge of their study system, and an understanding of the limitations and appropriate time and place to apply these methods. In Figure 6, we have attempted to summarize the sampling methodology most appropriate for four different study objectives.

Understanding how stream hiota respond across spatial sampling scales (reach or BVET) obviously is essential for properly designing field studies and monitoring programs (Ensign et al. 1997). For studies of population size or structure in small, species-poor streams, BVET methods (Hankin and Reeves 1988) likely are the best available choice. In these types of systems, the advantages of BVET methods (e.g., increased accuracy in population estimates, more statistically defensihle design) outweigh the disadvantages. BVET methods could be particularly well suited for detecting and monitoring species that tend to migrate seasonally up and down streams. Another potential application of BVET methods would be in measuring the effect of spatial configuration of hahitat units on fishes. For example, to determine whether riffles upstream of large pools have different fish characteristics than riffles upstream of small pools. An important caveat, however, is that BVET methods **will he** problematic to apply if stream size or species rich**ness** is too large, water clarity poor, or streams lack **hed** control structures.

If a comprehensive inventory of stream habitats (including location) is required to meet monitoring mandates (e.g., **hy** a USDA Forest Service Region) and the location of CGUs is important, BVET methods are the **hest** and perhaps only acceptable technique available. This assumes, however, that a repeatable system of CGU classification is used and the approach is only applied in temporally stable systems (especially in terms of flow regime and bed-control structures). A problem with large-scale habitat inventories is choosing streams that adequately represent conditions in the region. Few natural resource agencies have the funds, personnel, or time to conduct BVET surveys on every stream under their jurisdiction. The question then becomes, how many randomly chosen BVET surveyed streams would **he** needed to have confidence that they represent- conditions within a drainage or an entire national forest, for example? Other considerations when applying BVET methods to inventory habitats include the ability to accurately and precisely assign CGUs and the timing of sampling. Stream habitat can change dramatically over a matter of days in some streams (Williams et al. 2003a). If BVET methods are chosen for a particular study, we advocate using transects to characterize habitat in stream reaches that

Figure 6. Recommended sampling design and limitations for given study objectives in wadeable streams.



encompass a subset of all CGUs in the system (Toepfer et al. 2000, Vadas and Orth 1998) following a sampling design based on stream width (Simonson et al. 1994). Detailed data in all types of CGUs will allow more ease in scaling the data down from the whole-stream scale to individual reaches. The inability to reduce the spatial scale of data below that of the entire stream may be the biggest limitation of using the BVET methodology.

For many studies conducted in streams, the problems with BVET methods may be too great to overcome, limiting its valid application as a research or monitoring tool. In streams lacking bed control substrates (e.g., no bedrock, boulders, or large woody debris), classification of CGUs becomes extremely difficult, if not impossible. In many streams, snorkeling is not a viable sampling method because of high species diversity, high abundance, small size and cryptic coloration of many species, and turbidity. The conditions required for Hankin and Reeves' (1988) methods to be valid (Thompson 2003) simply cannot be met in these types of systems. Where BVET methods cannot be implemented, we recommend a statistically defensible, rigorously designed stratified reach or adaptive sampling (Smith et al. 2000) approach. Sampling 100 m of stream at accessible bridge crossings will not produce sufficient data to represent the habitat conditions and biota found in a system (Lyons 1992; Angermeier and Smogor 1995; Paller 1995; Patton et al. 2000). We recommend instead a stratified reach approach using transects and based on stream size (Simonson et al. 1994). Reaches should encompass all types of CGUs in a system (Lyons 1992; Angermeier and Smogor 1995; Paller 1995). A novel approach in these types of systems would be to use BVET methods to identify where to find the best representative reaches for a stream (Dolloff et al. 1997) and stratify reaches to encompass all types of CGUs. A more cost-effective alternative would be to walk the stream, visually identifying CGUs and georeferencing them prior to choosing reaches for sampling. Using data on available CGUs in a stream to stratify stream reaches would eliminate some of the problems and ensure that the reaches sampled are more representative of conditions found in the stream. Ideally, the precision of transect sampling could be coupled with some of the advantages of BVET methods, and sampled reaches would more accurately reflect conditions in the stream instead of the best access points.

Another key consideration of choosing between BVET and representative reach (or hybrid) approaches is to ensure that selected measures of population health are appropriate in the context of their intended use. Oftentimes, measures of environmental (e.g., in fisheries, forests, etc.) health are chosen with little regard for their intended use in monitoring or management efforts. As a result, many efforts by ecologists and resource managers aimed at identifying measures for environmental quality produce little more than long lists of potentially important biological or physical factors that are not useful from the standpoint of making management decisions. While this information is often science-relevant (in that it contributes to the growing body of knowledge about the ecology of a particular system), it is often not useful from the standpoint of specific resource management efforts that require measures that are sensitive to the geographic or event-specific concerns of agencies (Gregory et al. 2001, Gregory and Failing 2002).

For example, in cases where information about the response of fish populations to anthropogenic disturbance (e.g., logging impacts) is sought, dividing streams into various CGUs so that a BVET-based analysis may be undertaken is unlikely to be desirable (for the reasons noted above). While such an approach is likely to yield extensive data, other approaches (e.g., more intensive studies of certain fish populations in specific reaches or other more direct measures of the response of a system, which may be indirectly linked to population health), are likely to produce information that is more immediately useful for resource managers. In other cases—e.g., as part of intensive monitoring (vs. management) efforts designed to establish a detailed baseline-BVET reach approaches may be most appropriate, regardless of the level of effort required to complete the work. Making this distinction requires not only a detailed knowledge of the ecology of a given system, but also a clear understanding of why the study is being undertaken and *how* the data collected will be used by resource managers (Arvai and Gregory 2003).

In the end, how to go about determining the appropriate scale and measures to sample a stream system will continue to be an important question in the future (Labbe and Fausch 2000; Williams et al. 2002; Fausch et al. 2002). Perhaps the most important point to consider is that detailed data on factors influencing populations or assemblages collected at the wrong scale or with poorly defined management objectives can lead to spurious conclusions and poor management decisions (Arvai and Gregory 2003; Fausch et al. 2002). For this reason, we conclude with a caution: adopting one single method as the generic monitoring tool for streams without considering specific objectives of the study and limitations of the chosen method will degrade the quality of data, waste scarce research and monitoring dollars, and create unforeseen problems in management applications. Blind adoption of methods can lead to problematic management paradigms and limit the effectiveness of stream monitoring programs. While the recommendations provided in Figure 6 are by no means a comprehensive list of all possible study objectives or methodologies, it is our hope that these suggestions may prove useful to biologists and managers faced with the difficult task of assessing stream ecosystems.

Acknowledgments

We are grateful to J. A. Clingenpeel whose data from the Ouachita National Forest created the opportunity for this article. This article benefited from reviews by M. G. Williams, E. C. Braig IV, J. D. Hudson III, Z. Bowen, J. Peterson, and T. H. Bonner.

References

Angermeier, P. A., and R. A. Smogor. 1995. Estimating number of species and relative abundances in stream-fish communities: effects of sampling effort and discontinuous spatial distributions. Canadian Journal of Fisheries and Aquatic Sciences 52:936-949.

Arend, K. K. 1999. Macrohabitat identification. Pages 75-93 *in* M. B. Bain and N. J. Stevenson, eds. Aquatic habitat assessment: common methods. American Fisheries Society, Bethesda, MD.

Armitage, P. D., I. Pardo, and A. Brown. 1995. Temporal constancy of fauna1 assemblages in "mesohabitats"-application to management. Archiv fur Hydrobiologie 133:367-387.

Arvai, J. L., and R. Gregory. 2003. A decision focused approach for identifying cleanup priorities at contaminated sites. Environmental Science &Technology 37: 1469-1476.

Bisson, P. A., J. L. Nielsen, R. A. Pahnason, and L. E. Grove. 1982. A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low stream flow. Pages 62-73 in N. B. Armandtrout, ed. Acquisition and utilization of aquatic habitat information. Western Division of the American Fisheries Society, Portland, OR.

Cam, E., J. D. Nichols, J. R. Sauer, and J. E. Hines. 2002. On the estimation of species richness based on the accumulation of previously recorded species. Ecography 25:102-108.

Clingenpeel, J. A., and B. G. Cochran. 1992. Using physical, chemical and biological indicators to assess water quality on the Ouachita National Forest utilizing basin area stream survey methods. Proceeding of the Arkansas Academy of Science 46:33-35.

Cunjak, R. A., R. G. Randall, and E. M. I?. Chadwick. 1988. Snorkeling versus electrofishmg: a comparison of census techniques in Atlantic salmon rivers. Naturaliste Canadien 115:89-93.

D'Angelo, D. J., S. V. Gregory, L. R. Asbkenas, and J. L. Meyer. 1997. Physical and biological linkages within a stream geomorphic hierarchy: a modeling approach. Journal of the North American Benrhological Society 16:480-502.

Dolloff, C. A., D. G. Hankin, and G. H. Reeves. 1993. Basinwide estimation of habitat and fish populations in streams. USDA Forest Service, Southeastern Forest Experiment Station, General Technical Report, GTR-SE-83, Asheville, NC.

Dolloff, C. A., H. E. Jennings, and M. D. Gwen. 1997. A comparison of basinwide and representative reach habitat survey techniques in three southern Appalachian watersheds. North American Journal of Fisheries Management 17:339-347.

Ensign, W. E., P. L. Angermeier, and C. A. Dolloff. 1995. Use of line transect methods to estimate abundance of benthic stream fishes. Canadian Journal of Fisheries and Aquatic Sciences 52:213-222.

Ensign, W. E., K. N. Leftwich, P. L. Angermeier, and C. A. Dolloff. 1997. Factors influencing stream fish recovery following a large-scale disturbance. Transactions of the American Fisheries Society 126:895-907.

Fausch, K. D., C. E. Torgersen, C. V. Baxter, and H. W. Li. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. BioScience 52:483-498.

Frissell, C. A., W. J. Liss, C. E. Warren, and M. D. Hurley. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental Management 10:199-214.

Gregory, R., J. L. Arvai, and T. McDaniels. 2001. Value-focused thinking for environmental risk consultations. Research in Social Problems and Public Policy 9:249-275.

Gregory, R., and L. Failing. 2002. Using decision analysis to encourage sound deliberation: water use planning in British Columbia, Canada. Journal of Policy Analysis and Management 21:492-499.

Hankin, D. G., and G. H. Reeves. 1988. Estimating total fish abundance and total habitat area in small streams based on visual estimation methods. Canadian Journal of Fisheries and Aquatic Sciences 45:834-844.

Hawkins, C. P., J. L. Kershner, P. A. Bisson, M. D. Bryant, L. M. Decker, S. V. Gregory, D. A. McCullough, C. K. Overton, G. H. Reeves, R. J. Steedman, and M. K. Young. 1993. A hierarchical approach to classifying stream habitat features. Fisheries 18(6):3-12.

Hillman, T. W., J. W. Mullan, and J. S. Griffith. 1992. Accuracy of underwater counts of juvenile Chinook salmon, coho salmon, and steethead. North American Journal of Fisheries Management 12:598-603.

Kershner, J. L., and W. M. Snider. 1992. Importance of a habitat-level classification system to design instream flow studies. Pages 179-193 in P. J. Boon, P. Calow, and G. E. Petts, eds. River conservation and management. John Wiley & Sons, Ltd., Chichester, England.

Kolasa, J. 1989. Ecological systems in hierarchical perspective: breaks in community structure and other consequences. Ecology 70:36-47.

Labbe, T. R., and K. D. Fausch. 2000. Dynamics of intermittent stream habitat regulate persistence of a threatened fish at multiple scales. Ecological Applications 10:1774-1791.

Leftwich, K. N., P. L. Angermeier, and C. A. Dolloff. 1997. Factors influencing behavior and transferability of habitat models for a benthic stream fish. Transactions of the American Fisheries Society 126:725-734.

Lowe, W. H. 2002. Landscape-scale spatial population dynamics in human-impacted stream systems. Environmental Management 30:225-233.

Lyons, J. 1992. The length of stream to sample with a towed electrofishing unit when species richness is estimated. North American Journal of Fisheries Management 12:198-203.

Matthews, W. J. 1998. Patterns in freshwater fish ecology. Chapman & Hall, New York.

Matthews, W. J., B. C. Harvey, and M. E. Power. 1994. Spatial and temporal patterns in the fish assemblages of individual pools in a Midwestern stream (U. S. A.). Environmental Biology of Fishes 39:381-397.

McCain, M., D. Fuller, L. Decker, and K. Overton. 1990. Stream habitat classification and inventory procedures for northern California.

USDA Forest Service, Pacific Southwest Region, FHR Currents, San Francisco, CA.

Minshall, G. W. 1988. Stream ecosystem theory: a global perspective. Journal of the North American Benthological Society 7:263-288.

Paller, M. H. 1995. Relationship among number of fish species sampled, reach length surveyed, and sampling effort in South Carolina coastal plain streams. North American Journal of Fisheries Management 15:110-120.

Patton, T. M., W. A. Hubert, F. J. Rahel, and K. G. Gerow. 2000. Effort needed to estimate species richness in small streams on the Great Plains in Wyoming. North American Journal of Fisheries Management 20:394-398.

Peterson, J. T., and C. F. Rabeni. 2001. The relation of fish assemblages to channel units in an Ozark stream. Transactions of the American Fisheries Society 130:91 l-926.

Peterson, J. T., R. F. Thurow, and J. Guzevich. 2004. An evaluation of multi-pass electrofiihing for estimating the abundance of stream-dwelling salmonids. Transactions of the American Fisheries Society 133:462-475.

Platts, W. S., W. F. Megahan, and G. W. Minshall. 1983. Methods for evaluating stream, riparian, and biotic conditions. USDA Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report GTR-INT-138, Ogden, UT

Poole, G. C., C. A. Frissell, and S. C. Ralph. 1997. In-stream habitat unit classification: inadequacies for monitoring and some consequences for management. Journal of the American Water Resources Association 33:879-896.

Rabeni, C. F., K. E. Doisy, and D. L. Galat. 2002. Testing the biological basis of a stream habitat classification using benthic invertebrates. Ecological Applications 12:782-796.

Ralph, S. C., G. C. Poole, L. L. Conquest, and R. J. Naiman. 1994. Stream channel morphology and woody debris in logged and unlogged basins of Western Washington. Canadian Journal of Fisheries and Aquatic Sciences 51:37-51.

Roper, B. B., and D. L. Scamecchia. 1995. Observer variability in classifying habitat types in stream surveys. North American Journal of Fisheries Management 15:49-53.

Ross, S. T., M. T. O'Connell, D. M. Patrick, C. A. Latorre, W. T. Flack, J. G. Knight, and S. D. Wilkins. 2001. Stream erosion and densities of *Etheostoma rubrum* (Percidae) and associated riffle-inhabiting fishes-biotic stability in a variable habitat. Copeia 2001: 916-927.

Schlosser, I. J. 1991. Stream ecology: a landscape perspective. BioScience 41:704-712.

Schlosser, I. J. 1995. Critical landscape attributes that influence fish population dynamics in headwater streams. Hydrobiologia 303:71-81.

Sheldon, A. L. 1968. Species diversity and longitudinal succession in stream fishes. Ecology 49: 193- 198.

Simonson, T. D. 1993. Correspondence and relative precision of stream habitat features estimated at two spatial scales. Journal of Freshwater Ecology 8:363-373.

Simonson, T. D., J. Lyons, and l? D. Kanehl. 1994. Quantifying fish habitat in streams: transect spacing, sample size, and a proposed framework. North American Journal of Fisheries Management 14:607-615.

Smith, D. R., R. F. Villella, D. I? Lemarie, and S. von Oettingen. 2000. How much excavation is needed to monitor freshwater mussels? Proceedings of the First Freshwater Mollusk Conservation Society Symposium 1999:203-218.

Taylor, C. M. 1997. Fish species richness and incidence patterns in isolated and connected stream pools: effects of pool volume and spatial position. Oecologia 110:560-566.

Taylor, C. M., and M. L. Warren, Jr. 2001. Dynamics in species composition of stream fish assemblages: environmental variability and nested subsets. Ecology 82:2320-2330.

Thompson, W. L. 2003. Hankin and Reeves' approach to estimating fish abundance in small streams: limitations and alternatives. Transactions of the American Fisheries Society 132:69-75.

Thurow, R. F. 1994. Underwater methods for study of salmonids in the Intermountain West. USDA Forest Service, INT-GTR-307, Ogden, Utah.

Toepfer, C. S., W. L. Fisher, and W. D. Warde. 2000. A multistage approach to estimate fish abundance in streams using geographic information systems. North American Journal of Fisheries Management 20:634-645.

Vadas, R. L., and D. J. Orth. 1998. Use of physical variables to discriminate visually determined mesohabitat types in North American streams. Rivers 6:143-159.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37: 130-137.

Warren, M. L., Jr., and B. M. Burr. 1994. S&us of freshwater fishes of the United States: overview of an imperiledfauna. Fisheries 19(1):6-18.

Warren, M. L., Jr., and 11 coauthors. 2000. Diversity, distribution, and conservation status of the native freshwater fishes of the Southern United States. Fisheries 25(10):7-29.

Warren, M. L., Jr., W. R. Haag, and S. B. Adams. 2002. Forest linkages to diversity and abundance in lowland stream fish communities. Pages 168-182 *in* M. M. Holland, M. L. Warren, Jr. and J. A. Stanturf, eds. Sustainability of wetlands and water resources. USDA Forest Service, Southern Research Station, General Technical Report, GTR-SRS-50, Asheville, NC.

Williams, L. R., C. M. Taylor, M. L. Warren, Jr., and J. A. Clingenpeel. 2002. Large-scale effects of timber harvesting on stream systems in the Ouachita Mountains, Arkansas, USA. Environmental Management 29:76-87.

Williams, L. R., C. M. Taylor, and M. L. Warren, Jr. 2003a. Influence of fish predation on assemblage structure of macroinvertebrates in an intermittent stream. Transactions of the American Fisheries Society 132:120-130.

Williams, L. R., C. M. Taylor, M. L. Warren, Jr., and J. A. Clingenpeel. 2003b. Environmental variability, historical contingency, and the structure of regional fish and macroinvertebrate faunas in Ouachita Mountain stream systems. Environmental Biology of Fishes 67:203-216.