# BIOASSESSMENT OF SILVICULTURAL IMPACTS IN STREAMS AND WETLANDS OF THE EASTERN UNITED STATES

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**Abstract.** Bioassessment is a useful tool to determine the impact of logging practices on the biological integrity of streams and wetlands. Measuring biota directly has an intuitive appeal for impact assessment, and biota can be superior indicators to physical or chemical characteristics because they can reflect cumulative impacts over time. Logging can affect stream and wetland biota by increasing sedimentation rates, altering hydrologic, thermal, and chemical regimes, and changing the base of food webs. Biotic impacts of logging on streams compared to wetlands probably differ, and in this paper we review some of those differences. In streams, invertebrates, fishes, amphibians, algae, and macrophytes have been used as indicators of logging impacts. In wetlands, bioassessment is just beginning to be used, and plants and birds are the most promising indicator taxa. Various best management practices (BMPs) have been developed to reduce the impacts of logging on stream and wetland biota, and we review quantitative studies that have evaluated the efficacy of some of these techniques in streams and wetlands in the eastern United States. Remarkably few studies that address the overall efficacy of BMPs in limiting biotic changes in streams and wetlands after BMP implementation have been published in scientific journals, although some work exists in reports or is unpublished. We review these works, and compile conclusions about BMP efficacy for biota from this body of research.

**Keywords:** best management practices, BMP, forestry, logging, invertebrates, macrophytes, algae, fish, amphibians, reptiles, birds

## 1. Introduction

Silviculture in the eastern United States has had a pervasive influence on ecosystems for hundreds of years. Besides the obvious effects on terrestrial ecosystems, tree harvest, site preparation, and other silvicultural practices can greatly impact aquatic systems because of the tight linkages between streams, wetlands, and lakes and the watersheds they drain (e.g., Hynes, 1975; Bormann and Likens, 1979). Bioassessment, or using the organisms that live in aquatic environments as indicators of degradation, has become a standard practice throughout the US and the world. Bioassessment of silvicultural impacts has a particularly long history in eastern US streams (Tebo, 1955) and has become an essential part of stream monitoring programs for most states (Davis *et al.*, 1996). The growing realization that maintaining water quality (i.e., water chemistry) standards alone does

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not necessarily protect biological integrity (Karr and Chu, 1999) has led to the proliferation of stream bioassessment programs. The successful implementation of bioassessment for streams and the growing recognition of the value of wetlands have led researchers to apply bioassessment to wetlands over the last 10 years (Rader *et al.*, 2001).

Since the passing and legal enforcement of the Clean Water Act, point-source pollution (i.e., that from a defined location such as a pipe) has been ameliorated greatly. Today, however, aquatic systems primarily face different threats, e.g., non-point-source pollution produced from land use practices such as agriculture, silviculture, and urban and suburban development. Mitigation of detrimental effects associated with non-point-source pollution has come primarily in the form of best management practices (BMPs) that are designed to reduce the amount of on-site damage to habitats, as well as limit off-site effects due to the export of harmful materials (e.g., sediment, toxics). BMPs are implemented widely across the US as part of forestry activities.

In this paper, we review the literature on the use of biota to assess the impacts of timber harvest and other silvicultural practices on streams and wetlands of eastern North America. The results of studies published in peer-reviewed journals are the focus of our review because they represent the 'state of the art'. We first review how stream and wetland biota are influenced by various harvest and site preparation practices used for silviculture in eastern North America. We then determine which organisms are preferentially used for bioassessment in streams and wetlands. We finally assess the efficacy of certain silvicultural practices, especially BMPs, in protecting the biotic integrity of streams and wetlands. One goal of this paper is to highlight how bioassessment protocols should probably differ between streams and wetlands due to their respective unique characteristics.

## 2. Silvicultural Influences on Stream and Wetland Biota

### 2.1. Streams

Stream flow is elevated after tree removal due to reduced evapotranspiration and interception (e.g., Hornbeck *et al.*, 1978; Verry, 1986; Swank *et al.*, 1988). Typically, both baseflow and peak stormflow are increased. Long-term (> 20 yr) studies of clear-cutting in the Hubbard Brook Experimental Forest (Martin *et al.*, 2000) and the Coweeta Hydrologic Laboratory (Swank *et al.*, 2001) report that stream hydrologic characteristics return to pre-treatment levels after 5 years as early successional vegetation rapidly restores rates of evapotranspiration and interception. We are unaware of any studies in the eastern US showing that stream organisms are directly impacted by logging-induced changes in hydrology. Instead, biota are influenced more by increases in erosion and sedimentation, which can be partly due to increased stream flow.

Fine sediments are responsible for the majority of impaired stream miles in the U.S. (US EPA, 2000), and one source of increased sediments to streams is forestry activities. Two percent of all assessed stream miles (7% of all impaired miles) were reported to be degraded by forestry (US EPA, 2000). Sediments can negatively impact stream biota in a number of ways (see reviews by Waters, 1995 and Wood and Armitage, 1997). Waters (1995) details the sources and impacts of sediment from forestry activities. Erosion from multiple sources (i.e., within the stream channel due to increased peakflow, in the watershed due to site preparation activities and logging roads during construction and later use) mobilizes and enhances sediment inputs to streams, although logging roads are, by far, the largest source of sediments to streams.

Numerous studies, especially the long-term study of Big Hurricane Branch at Coweeta (see reviews by Wallace, 1988 and Webster *et al.*, 1992), have examined how sediment from logging activities impacts stream invertebrates. Gurtz and Wallace (1984) found that abundance of many invertebrate taxa in habitats susceptible to sediment deposition (i.e., pools and sandy reaches) declined in a stream draining a recently clear-cut watershed, whereas those taxa in less susceptible habitats (i.e., steep-gradient, boulder outcrops) increased. This trend was still evident five years later (Wallace *et al.*, 1988), but not after 16 years (Stone and Wallace, 1998). This pattern in invertebrate response corresponded to a lack of new sediment inputs to the stream after road construction and the routing of the initial sediment pulse through the stream system (Swank *et al.*, 2001).

Sedimentation did not appear to have a major, long-term impact on macroinvertebrates in streams draining watersheds that had been logged for many years before the studies took place (Silsbee and Larson, 1983; Williams *et al.*, 2002), or in a low-gradient stream where invertebrates are better adapted to fine sediments than in high-gradient streams (Kedzierski and Smock, 2001). Species richness and diversity of certain invertebrates in a southern Appalachian stream were impacted by sedimentation from logging operations and residential construction and nutrient enrichment from nearby livestock pastures (Lemly, 1982).

Studies focusing on other taxa have also found that sedimentation from logging can influence biota. Sediment effects on fishes, especially their reproductive success, have been well studied in the western U.S. (Waters, 1995), but not as much in the eastern U.S. Jones *et al.* (1999) found removal of riparian forests in the southern Appalachians was associated with increased sediment deposition, which altered fish assemblage structure to favor species that were sediment tolerant. However, the tree removal was also associated with some agricultural activity that may be responsible for the changes. Fish in streams of the Ouachita Mountains did not respond to timber harvesting (Williams *et al.*, 2002). A salamander, *Gyrinophilus porphyrtiticus*, was less abundant in streams with high substrate embeddedness due to sediment input from logging (Lowe and Bolger, 2002).

An obvious effect of logging is an increase in insolation to the stream before growth of early successional vegetation shades the stream. Increased sunlight can cause an increase in stream temperature (e.g., Swift and Messer, 1971), as well as boost primary production. Warmer temperatures in a stream draining a clearcut watershed allowed some downstream invertebrates that preferred warmer water to move up into the usually cooler headwaters (Stone and Wallace, 1998). Lowe *et al.* (1986) found more algal biomass in a southern Appalachian stream draining a clear-cut watershed, as well as altered community structure and growth form because algae were released from light limitation. Periphyton growth was also higher in New England streams draining 2- and 3-year-old clearcuts (Noel *et al.*, 1986). Algal biomass increased during the summer in streams draining harvested areas of Hubbbard Brook (Ulrich *et al.*, 1993). Increased light also greatly stimulated growth of two macrophytes in a low-gradient Coastal Plain stream (Kedzierski and Smock, 2001).

These increases in primary producers in turn can enhance both habitat and food resources for stream macroinvertebrates. The additional macrophyte biomass reported by Kedzierski and Smock (2001) provided increased surface area for filter-feeding macroinvertebrates, which allowed abundance and production of these filterers to be greatly enhanced compared to an unlogged section of stream. Noel *et al.* (1986) found higher abundance of invertebrates associated with the increased periphyton growth. The long-term studies of Big Hurricane Branch have found increased abundance, biomass, and production of grazing invertebrates, especially the mayfly *Baetis* (Gurtz and Wallace, 1984; Wallace and Gurtz, 1986). However, *Baetis* production declined over 16 years with the return of shading from the growing forest (Wallace *et al.*, 1988; Stone and Wallace, 1998).

The switch from allochthonous to autochthonous food resources in eastern streams draining logged watersheds is relatively short-lived (ca. 5 years) due to the rapid regrowth of terrestrial vegetation (Webster *et al.*, 1992). However, a longer-term effect on stream biota is the change in leaf quality entering streams. Early-successional leaf species tend to break down faster in streams than late-successional species and may be better food resources for leaf-shredding invertebrates (Webster *et al.*, 1983; Benfield *et al.*, 1991). This change in leaf quality partly explains why the abundance, biomass, and productivity of detritivorous invertebrates, especially shredders, are often higher for many years after clearcutting (Silsbee and Larson, 1983; Gurtz and Wallace, 1984; Wallace *et al.*, 1988; Griffith and Perry, 1991; Stout *et al.*, 1993; Stone and Wallace, 1998). Further, these increases in shredders contribute to increased breakdown rates of all leaf species in streams draining $>1$-year-old logged watersheds (Webster and Waide, 1982; Benfield *et al.*, 1991; Griffith and Perry, 1991; Benfield et al., 2001).

An additional factor contributing to the higher leaf breakdown is higher concentrations of nutrients, especially nitrate, in streams following logging (Bormann *et al.*, 1974; Likens *et al.*, 1978; Swank, 1988; Martin et al., 2000; Swank *et al.*, 2001). This nutrient increase may stimulate microbial activity on leaf surfaces, although this has not been tested for logging-impacted streams. However, the increase in nitrate concentration is typically short-lived because of forest regrowth, unless species-specific successional dynamics (e.g., mortality of early successional nitrogen-fixing species such as *Robinia pseudoacacia* L.) influence nutrient dynamics (Swank *et al.*, 2001).

The final major effect of logging on streams is the loss of coarse woody debris input. Because logging removes large trees that would eventually fall into streams, streams draining logged forests have less woody debris than those draining undisturbed forests (Silsbee and Larson, 1983; Flebbe and Dolloff, 1995; Hedman *et al.*, 1996; Flebbe, 1999). This loss of wood leads to a loss of habitat and cover, which in turn negatively impacts certain fishes (Angermeier and Karr, 1984; Flebbe and Dolloff, 1995; Flebbe, 1999). Experimental additions of large wood in streams also result in changes in benthic macroinvertebrate assemblages (Wallace *et al.*, 1995; Hilderbrand et al., 1997), so it is likely that less wood in logged streams also influences invertebrates through changes in habitat and food resources. In addition to effects on channel geomorphology, large wood forms the basis of debris dams in streams, and streams draining logged watersheds in the east have fewer debris dams (Hedin *et al.*, 1988; Golladay *et al.*, 1989). Debris dams greatly influence retention of organic matter (i.e., food resources for invertebrates) and sediment in streams (Bilby and Likens, 1980; Bilby, 1981; Wallace *et al.*, 1995).

### 2.2. Wetlands

Like streams, the hydrology of wetlands can be altered by timber harvest and other silvicultural practices (see reviews by Richardson and McCarthy, 1994; Lockaby *et al.*, 1997a; Sun *et al.*, 2001). In general, transpiration and interception rates decline after harvest and wetlands become ‘wetter’ in the years immediately after harvest, although both seasonal variation in weather (Sun *et al.*, 2001) and soils (Lockaby *et al.*, 1994) can alter this kind of response. Road building, culverts, and ditches can alter velocity and direction of water flows, affecting overall wetland hydrology. While most wetland organisms are strongly influenced by wetland hydrology (Batzer and Wissinger, 1996, Mitsch and Gosselink, 2000), alteration of hydrology induced by tree harvest has rarely been implicated as the reason for biotic change in wetlands. Batzer *et al.* (2000) suggest that, at least in highly variable wetlands, hydrologic alterations induced by logging can be dwarfed by natural annual or seasonal variation.

Perhaps the most dramatic impact of timber harvest on wetlands is the associated change in plant communities. After harvest, many formerly forested wetlands take on the floristic characteristics of marshes (Perison *et al.*, 1997; Gale *et al.*, 1998; Batzer *et al.*, 2000) or in some cases develop an upland flora (Mitchell *et al.*, 1995; Roy *et al.*, 2000). Aust *et al.* (1997) found that applying herbicides after harvest can accentuate the development of marshy conditions. The nature of the plant community can dictate the structure of the animal communities (Wigley and Roberts, 1994a). Although a diversity of animals live in both forested and harvested wetlands, the species composition often shifts from a forested fauna to a fauna more typical of marshes and meadows (Hurst and Bourland, 1996; Clawson *et al.*, 1997; Perison *et al.*, 1997; Batzer *et al.*, 2000; Moorman and Guynn, 2001; Prenger and Crisman, 2001) or uplands (Mitchell *et al.*, 1995; Phelps and Lancia, 1995; Perison *et al.*, 1997). Moorman and Guynn (2001) found that harvest increased the numbers of edge-inhabiting birds. Arboreal reptile populations are negatively impacted by clear cutting (Enge and Marion, 1986), while reptiles associated with open areas and groundcover may benefit (Phelps and Lancia, 1995; Perison *et al.*, 1997). Fragmentation of wetland forests may also influence bird populations on a regional level (Wigley and Roberts, 1997).

The wet soils in wetlands are vulnerable to compaction and rutting if heavy equipment is used for harvest (Aust *et al.*, 1993). Substrate texture and slowed water velocity resulting from slash, rutting, and post-harvest growth of herbaceous plants can trap suspended sediments in wetlands (Johnston, 1991; Rapp *et al.*, 1999). Increased levels of solar radiation following canopy loss can affect water and soil temperatures and potentially amphibians (Clawson *et al.*, 1997).

### 2.3. Differences between stream and wetland systems

In terms of timber harvest impacts, a major difference between streams and most wetlands is that streams are affected by impacts on neighboring upland or riparian zones while impacts on wetlands can often be direct. While trees have an important influence on streams, trees are clearly the keystone organisms of forested wetlands.

Issues of sediment probably also differ between streams and wetlands. Perhaps the most important impact on streams of tree harvest and associated road building and site preparation is the import of sediment from adjacent uplands. For wetlands, silvicultural practices may not increase sediment import because much of the activity is on site. Even for isolated wetlands set in harvested tracts, sediment import might not be a major factor because at least in the eastern US most of these wetlands are located in flat areas where surface runoff is unlikely to occur (Batzer *et al.*, 2000). However, sediment issues remain important for wetlands because sediments mobilized from harvest might enter and impact downstream channels and associated wetlands. Further, slash and herbaceous growth in harvested wetland tracts may trap more sediment even if import levels are not altered (Johnston, 1991).

## 3. Organisms Used for Bioassessment

### 3.1. Streams

Several taxonomic groups have been used for assessing in-stream effects of logging in the east, but macroinvertebrates have been used far more often than algae, macrophytes, amphibians, and fishes. We found many papers (Woodall and Wallace, 1972; Lemly, 1982; Silsbee and Larson, 1983; Gurtz and Wallace, 1984, 1986; Wallace and Gurtz, 1986; Wallace *et al.*, 1988; Griffith and Perry, 1991; Stout *et al.*, 1993; Adams *et al.*, 1995; Brown *et al.*, 1997; Hutchens *et al.*, 1997; Stone and Wallace, 1998; Kedzierski and Smock, 2001; Vowell, 2001; Williams *et al.*, 2002) directly examining macroinvertebrates in logging impacted streams. Clearly, macroinvertebrates are useful for showing how a watershed disturbance like logging affects stream biota. We found surprisingly few papers on how logging influences fishes (Rutherford *et al.*, 1992; Flebbe and Dolloff, 1995; Flebbe, 1999; Jones *et al.*, 1999; Williams *et al.*, 2002), especially when compared to the rich literature in the western US focused on salmonids. Algae were the center of three studies (Lowe *et al.*, 1986; Noel *et al.*, 1986; Ulrich *et al.*, 1993) on logging effects, and salamanders were the center of two (Stiven and Bruce, 1988; Lowe and Bolger, 2002). Macrophytes were examined in one study (Kedzierski and Smock, 2001), and may be particularly useful in low-gradient streams where macrophytes are more common.

In this review we focused on logging effects on in-stream biota. However, there is also a large literature associated with logging effects on organisms living in the riparian zone (see review by Verry *et al.*, 2000). Studies and reviews examining logging effects in the riparian zone have used birds (e.g., DeGraaf and Yamasaki, 2000; Hanowski *et al.*, 2003), mammals (e.g., DeGraaf and Yamasaki, 2000; Ford and Rodrigue, 2001), amphibians and reptiles (e.g., Pauley *et al.*, 2000).

### 3.2. Wetlands

We found 25 published research papers that used biotic data to assess impacts of silvicultural practices on wetlands of the eastern U.S. or used biota to assess recovery of regional wetlands to harvest. By far the organisms most commonly used for wetland bioassessment were plants (12 of 25 studies) (Pavel and Kellison, 1992; Hauser *et al.*, 1993; Ewel, 1996; Aust *et al.*, 1997; Lockaby *et al.*, 1997b; Messina *et al.*, 1997; Perison *et al.*, 1997; Gale *et al.*, 1998; Rapp *et al.*, 1999; Batzer *et al.*, 2000; Roy *et al.*, 2000; Palik *et al.*, 2001). The use of plants, especially trees, is logical for wetland forests because the trees themselves are being harvested and monitoring the successful regeneration of trees should indicate habitat recovery. Besides tree regeneration, general floristics is also commonly monitored (8 of those 12 studies).

The next most commonly used wetland indicator organisms were amphibians with 7 published research studies (Enge and Marion, 1986; Phelps and Lancia, 1995; Clawson *et al.*, 1997; Perison *et al.*, 1997; Palik *et al.*, 2001; Russell *et al.*, 2002; Ryan *et al.*, 2002). Birds are commonly touted as being useful bioindicators of upland forest fragmentation (e.g., Petit *et al.*, 1995), and the theory for using them in assessing the biotic integrity of wetland forests is being developed (Wigley and Roberts, 1994a, b, 1997). However, theory has not been widely put into practice in eastern North America; we found only 3 research articles that used wetland bird data to assess impacts of timber harvest (Hurst and Bourland, 1996; Twedt *et*

TABLE I

Types of organisms used for bioassessment of silvicultural impacts in streams and wetlands of the eastern US, ranked from the most commonly used to the least

| Streams | Wetlands |
|---|---|
| 1. Invertebrates | 1. Macrophytes |
| 2. Fishes | 2. Amphibians |
| 3. Algae | 3. Birds |
| 4. Amphibians | 4. Invertebrates |
| 5. Macrophytes | 5. Mammals |
| 6. Birds | 6. Algae |
| 6. Mammals | 6. Fishes |

*al.*, 1999; Moorman and Guynn, 2001). Numerous reports, M.S. theses, and Ph.D. dissertations exist that use birds for bioassessment of wetland harvest, but until this approach is scrutinized by peer review its efficacy remains unsubstantiated.

While invertebrates are commonly used in stream bioassessment, we found only 2 research papers (Batzer *et al.*, 2000; Palik *et al.*, 2001) and a book chapter (Prenger and Crisman, 2001) that used wetland invertebrate data for bioassessment of timber harvest. A single research paper was found that used small wetland mammals as bioindicators of tree harvest (Mitchell *et al.*, 1995), and none were found that used fish or algae.

### 3.3. Differences between streams and wetlands in the organisms used for bioassessment

In terms of use by practitioners, we rank in Table I the relative importance of macrophytes, algae, invertebrate, fish, amphibians, birds, and mammals for bioassessment in streams and wetlands in the eastern U.S. While invertebrates are the most frequently used organisms for bioassessment of silvicultural impacts in streams, they have not been widely used for wetlands. At this point, the minimal use of invertebrates for wetland bioassessment simply reflects a lack of effort. However, wetland habitats can naturally be stressful, so wetland invertebrates may be more tolerant (Batzer and Wissinger, 1996) of harvest-induced stress and less useful as indicators than invertebrates in streams. Macrophytes (including trees) are the most widely used biotic measure to assess impacts of tree harvest on wetlands, whereas they are not used as much in streams. Bioassessment of silviculture in streams and wetlands apparently requires habitat-specific protocols.

## 4. Efficacy of BMPs

### 4.1. Streams

Most BMPs for protecting streams from logging center on using harvesting and road building practices to reduce sources of sediment input and on retaining a buffer strip of riparian trees to filter water, maintain shading, and protect leaf and wood inputs. Assessing the effectiveness of BMPs has been limited, with most studies addressing the protection of water quality (e.g., temperature, sediment and nutrient load) rather than biotic resources. In a review of early BMP studies, Corbett *et al.* (1978) concluded most BMPs were effective for reducing logging-related problems with water quality. A number of effective ways to reduce sediment erosion from logging roads through proper construction and maintenance techniques were shown by Swift (1988). Arthur *et al.* (1998) found that BMP implementation in an eastern Kentucky watershed moderately reduced sediment load and nitrate concentrations. In an area of Mississippi with more erodible soils, Keim and Shoenholtz (1999) reported that maintenance of riparian buffer strips reduced total suspended solids (TSS) because they eliminated soil disturbance near the stream rather than filtering sediments from overland flow. Wynn *et al.* (2000) also found that TSS were not increased during logging when BMPs were implemented in a coastal watershed in Virginia.

We found only two studies that examined BMP effectiveness with respect to biota, and both of these examined benthic macroinvertebrates. Adams *et al.* (1995) studied whether BMPs were effective for reducing impacts on stream habitat and macroinvertebrates at 27 sites in five physiographic regions South Carolina. They used rapid bioassessment protocols III (RBPs) established by the US EPA (Plafkin *et al.*, 1989) as their monitoring tool. Adams *et al.* (1995) found that most sites using BMPs scored well on their RBP scores, and were thus effective in protecting macroinvertebrate assemblages. Vowell (2001) examined BMP effectiveness in northern Florida using the Stream Condition Index (SCI) developed specifically for Florida. Similar to the RBP used by Adams *et al.* (1995), the SCI incorporates evaluations of both habitat and stream macroinvertebrates. Four sites were studied before and after silvicultural treatments, in which all applicable BMPs were followed. Vowell found no effect of logging on SCI scores, supporting the value of BMPs for protecting stream macroinvertebrates. Although the results of these two studies support the use of BMPs, both studies could have been improved greatly by including logged sites in which BMPs were not used. Hence, additional studies examining multiple biotic assemblages with all potential treatments and controls included are still needed to better assess the efficacy of BMPs.

## 4.2. WETLANDS

Management strategies to reduce impacts of silviculture on wetland biota will differ depending on whether the wetland itself is harvested (e.g., bottomland hardwood forests) or whether the wetland is simply located in a harvested landscape (e.g., depressional wetlands set in an upland forest). In wetlands that are directly harvested, the trees themselves are the keystone taxa for the ecosystems and most biotic change appears to result from the loss of the trees (Clawson *et al.*, 1997; Perison *et al.*, 1997; Twedt *et al.*, 1999; Moorman and Guynn, 2001; Prenger and Crisman, 2001). Some biotic change is probably inevitable following harvest, but permanent biotic change can be avoided if wetland forests successfully regenerate. Ewel (1996) suggests that cutting pondcypress trees in winter and leaving low stumps (< 70 cm) may enhance regeneration from sprouting. Avoiding post-harvest applications of herbicide may enhance regeneration of tupelo, cypress, and ash trees (Aust *et al.*, 1997). These authors did not, however, find that harvest via helicopter enhanced regeneration rates compared to skidder harvest (in fact the opposite pattern was observed). Harvest technique (Pavel and Kellison, 1992) and site preparation (windrowing, secondary ditching) (Hauser *et al.*, 1993) can influence how post-harvest plant communities develop. Messina *et al.* (1997) found that natural wetland forests began developing in bottomland hardwood sites that were harvested using systems that did not rely on site preparations. Moorman and Guynn (2001) recommend retaining patches of mature forest in harvested areas that are of sufficient size to serve as breeding habitat for forest-interior birds.

Some silvicultural practices may directly affect wetland animals. It is generally acknowledged that operating heavy equipment can harm animals living on-site. Enge and Marion (1986) report that minimizing site preparations can reduce or eliminate impacts on herpetofauna; they further report that leaving residual logging debris benefits reptiles. However, most reported influences on wetland animal populations appear to be linked with changes in flora rather than the direct impact of any operation.

For small depressional wetlands, Prenger and Crisman (2001) indicate that direct harvest will induce many of the same biotic changes that occur when larger wetland forests are harvested. However, impacts of silvicultural practices on biota of small depressional wetlands appear to be minimal if harvest and site preparation activities are restricted to the adjacent uplands (Russell *et al.*, 2002). Batzer *et al.* (2000) report that peripheral harvest can induce the development of marshy conditions (as does direct harvest, see above), although negative impacts on invertebrates were not detected. Palik *et al.* (2001) report that plant and animal communities in depressional ponds were similar in ponds set in recently harvested (< 7 years) landscapes compared to those set in mature (> 75 years regrowth) forest. Semlitsch (1998) clearly established the ecological importance of surrounding uplands to pond salamanders, and Russell *et al.* (2002a) suggested that herpetofauna using both small depressional wetlands and adjacent uplands can be influenced by management in adjacent uplands. However, in some South Carolina depressional wetlands, silvicultural activity in adjacent uplands had minimal negative impacts on amphibians and reptiles (Russell *et al.*, 2002b). These authors suggest that the need for upland forest buffers around wetlands will depend on landscape features and the composition of the resident fauna.

The general efficacy of BMPs for wetlands has not yet been empirically tested using biota. Because most responses are linked to the loss of trees, it seems unlikely that any silvicultural practice will eliminate impacts on other biota. Best management practices that focus on promoting the regeneration of natural wetland forests should minimize any long-term impacts on wetland biota (Messina *et al.*, 1997).

## 5. Conclusions and Future Directions

### 5.1. Streams

Although studies of forestry practices on biota have a longer history in streams than in wetlands, there is still much room for further research in streams. We need well-designed studies focused on BMP effectiveness for multiple taxonomic groups throughout the eastern US. Furthermore, studies that examine the utility of individual BMPs would help determine the relative effectiveness of different BMPs.

One major research gap is the appropriate riparian buffer widths for a variety of forest types and regions. Most buffer width studies have focused on lessening the effects of agriculture (e.g., Lowrance *et al.*, 1991; Osborne and Kovacic, 1993; Castelle *et al.*, 1994), and it is unclear whether similar buffer widths are required for different types of land use in different physiographic regions. We know little about how natural variation (e.g., slope, soil type, stream width, tree height) influences the width needed to protect both water quality and stream biota. In addition, we need studies that examine the length of upstream strips of riparian vegetation (e.g., Harding *et al.*, 1998; Jones *et al.*, 1999; Sponseller *et al.*, 2001) as well as the width if we are to better understand and manage our forests.

The final consideration with respect to logging impacts on biota is an appreciation of temporal scale. Much of the variability seen in the different studies reviewed here relates to the timing of the study with respect to when the logging occurred. Many of the obvious effects of logging on water quality, stream habitat, and stream biota are relatively short-lived (i.e., $< 5$ years) because eastern US forests grow fairly rapidly. However, there are some subtler, long-lasting effects with respect to leaf quality and woody debris dynamics. In order to completely understand the effects of logging, researchers should examine recovery over longer time frames than are typically done (e.g., Stone and Wallace, 1998). In addition, we should be cognizant of how past land use influences our view of current stream structure and function. The eastern US has a long history of land use (Benfield,

1995) whose effects may be difficult to discern in the landscape of today (Harding *et al.*, 1998). We should also ask whether our current view of the impact of logging might be affected by the fact that most of our study streams are set in second- and third-growth forests.

### 5.2. Wetlands

For wetlands of eastern North America, we found that bioassessment has been effectively used to assess wide reaching impacts of various silvicultural practices. Responses of plants have been reasonably well documented. Published studies on responses to harvest by most animals are lacking, however. Much of the evidence that we reviewed suggests that efficient regeneration of pre-existing wetland forests should result in the recovery of most other wetland plants and animals. However, that hypothesis has not been adequately tested. Most research studies assess short-term responses only. Studies assessing longer-term responses of biota tend to be retrospective (Batzer *et al.*, 2000; Palik *et al.*, 2001); while these kinds of correlative studies remain valuable, experimental approaches would be more rigorous (e.g., Aust *et al.*, 1997; Rapp *et al.*, 1999). The experimental approaches used to assess short-term responses need to be extended into the long-term to verify that wetland forests regenerate and whole wetland communities recover from harvest.

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