Ouachita and Ozark Mountains Symposium:
Ecosystem Management Research

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Symposium objectives:

- to present an overview of ecosystem management research in the Ouachita and Ozark Mountains
- to present results from 5 years of postharvest measurements from the phase II stand-level studies including silviculture, understory vegetation, wildlife, soils and water quality; logging and economics; visual quality; and arthropods and microbial diversity
- to present results from 5 years of baseline data collection from the phase III landscape-scale studies including vegetation, wildlife, aquatic ecology, hydrology, and social sciences.
- to present an overview of other ecosystem-based projects in the Interior Highlands.

Symposium sponsors:

U.S. Department of Agriculture (USDA), Forest Service, Southern Research Station
University of Arkansas School of Forest Resources, Arkansas Forest Resources Center
USDA Forest Service, Ouachita National Forest
USDA Forest Service, Ozark-St. Francis National Forest

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Opening Session

James B. Baker, Moderator

THE HISTORY OF NEW PERSPECTIVES AND ECOSYSTEM MANAGEMENT

F. Dale Robertson[1]

Abstract—Arkansas occupies a unique and important place in the history of New Perspectives and Ecosystem Management. An historic visit to the Ouachita National Forest by Senator David Pryor (D-Arkansas) in August 1990, thereafter called the walk in the woods, served as an opportunity to shift the Ouachita's style of management in a manner that has served as a model for other national forests in the Nation. This paper summarizes limits of the multiple-use concept, outlines some important elements that constitute the basis of the New Perspectives program, and discusses from the author's perspective the evolution of the Ecosystem Management concept while he served as Chief of the USDA Forest Service.

INTRODUCTION

Thanks for this opportunity to come back to my home State, and to the Ouachita National Forest (NF), where I began my Forest Service career. Dr. Jim Baker, a key leader of the research underlying this symposium, and I were classmates at Arkansas Agricultural and Mechanical (A&M) College, now the University of Arkansas at Monticello (UAM). He went on to become a world-class research scientist. I pursued a less honorable career path and became Chief of the Forest Service, and by doing so got an opportunity for early retirement—which is an occupational hazard for Chiefs!

This morning, I was asked to kick off your symposium by talking about the history of the New Perspectives and Ecosystem Management concepts, the importance of the Ouachita NF in that history, and my view of changes that have occurred in the last 9 years. I am eager to hear and see first-hand how things are going 9 years after the so-called walk in the woods back in August 1990 with Senator David Pryor. Tomorrow, I'm taking another walk in the woods, but the good news is that I have absolutely no authority to make any decisions. So you Rangers can relax, you do not have anything to worry about!

I will give you my perspective as Chief from the Washington, D.C., vantage point. National policy making is never a very pretty thing, even for a highly professional outfit like the Forest Service, when it involves a scientific topic like Ecosystem Management. It was also a sensitive political issue that had to play out in the Washington, D.C., political environment. So, here is my story.

THE LIMITS OF MULTIPLE-USE MANAGEMENT

Fortunately or unfortunately during my tenure as Chief, multiple-use management, as it was being practiced on the national forests, hit a wall. Management of the national forests got mired down in intense public controversy, with many of our land management decisions being appealed and challenged in the courts. It was taking an extraordinary amount of time and effort to get even the simplest jobs done, like timber sales here on the Ouachita NF.

For the first time in its history, the Forest Service began to fall short in a major way of meeting its financed goals and targets as outlined in the congressionally approved budget and appropriation laws. The Forest Service had always been known as a can-do outfit. I was rapidly becoming the most experienced Chief in the history of the Forest Service in trying to explain to Congress and user groups why the Forest Service was not meeting its financed goals and targets. I think that probably everyone here pretty well knows the reasons behind the Forest Service's situation at that time, so I won't spend much time on that.

Even though clearcutting was really a big issue here in Arkansas, it was not a new issue. In fact, the first professionally trained forester in America, Bernard E. Fernow, was fired in 1903 for clearcutting on the Cornell University Experimental Forest in the Adirondack Mountains of New York. Due to the clearcutting issue, he completely lost the first forestry school in America when New York State decided to zero out his budget.

Thus, clearcutting has always been a threatening cloud hanging over the forestry profession from the very beginning of American forestry. The American people simply do not like the way clearcuts look, and they highly value the scenic qualities of their national forests. That is something that Forest Service people in Arkansas know as well or better than anyone else in the country.

The primary new drivers of change in management of the national forests were the Endangered Species Act (ESA), the viability requirement of the National Forest Management Act (NFMA), the National Environmental Policy Act (NEPA), and the courts. The ESA was the most difficult to deal with because it really established the protection of critical habitat for endangered species as a dominant use. The Forest Service has had great difficulty fitting endangered species, such as the northern spotted owl, within our overall multiple-use management concept.

Over the years, under the multiple-use concept the Forest Service had gotten very good and comfortable at considering the relative values of many resources, making trade-

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offs, and arriving at what we thought were good, balanced decisions. This was the trademark that the Forest Service had been well known for since the days of Gifford Pinchot. But, when we got hauled into court, most of the judges either did not understand our decision-making process, thought it was procedurally inadequate under NEPA, or disagreed with the decision. The trade-offs that we had been making over the years under the concept of multiple-use became no longer acceptable. And, the Forest Service was fast building up a bookcase full of case law that made it increasingly difficult to manage the national forests under the thinking of that time.

So, the time was ripe for some rather drastic conceptual changes in how the Forest Service managed the national forests. And I happened to be the Chief at the time serving in a Republican Administration that was not too keen on making major policy changes, especially if it would adversely affect economic benefits from the national forests.

CHANGES IN THINKING AND POLICY

It was relatively easy to conclude that the concept of multiple use, especially with a heavy reliance on clearcutting, had its shortcomings. As it was being practiced, the concept was not comprehensive enough to encompass all the considerations and values that the Forest Service had to take into account in the future management of the national forests. But few people had better alternatives. At the time, the choice was either to continue to shore up and strengthen the multiple-use management concept, or to make a major conceptual change in our way of thinking about how the national forests should be managed.

Well, we decided to start down the path of change, not knowing for sure where it was going to lead us. At the time, the term ecosystem management was not a common term in our vocabulary—maybe only in the minds of a few of our ecologists and other scientists.

New Forestry

Jerry Franklin, a research forest ecologist at the Pacific Northwest Forest and Range Experiment Station, was one of the first to take an ecological approach in dealing with the old growth/spotted owl issue. As I remember, Jerry's new idea was then that we should view the forest as an ecosystem and expand our thinking and planning to encompass something called a landscape. An ecosystem at the landscape level had certain structural characteristics like old-growth trees, large snags, down logs, and shaded riparian areas. Jerry recommended that all of these ecological values should be perpetuated in the forest through time in order to maintain all pieces of the ecosystem (including endangered species). At about the same time, a few other Forest Service scientists, like Tom Crow in the Lake States, were also developing similar ecological concepts.

Jerry began to work with district rangers in the Pacific Northwest to pilot test his ecological concepts in the design of timber sales. Soon we were able to see what Jerry's ideas looked like on the ground. As Chief, I went out to see some of the results. It was certainly different and in many ways looked like a sloppy, unfinished logging job!

Perhaps unfortunately, Jerry called it new forestry. It generated a lot of controversy in the Pacific Northwest. Some old timers called it old forestry, referring to the kind of high-grade logging that they did back in the 1950's. As you might suspect, Jerry was not very popular with many in the forestry community of the Pacific Northwest. But, as Chief I knew Jerry personally from my days as Forest Supervisor of the Siuslaw NF and had a lot of confidence in his work. Besides, no one else was coming forth with any better ideas about how to solve our old-growth/spotted owl issue in the Pacific Northwest.

I ended up giving Jerry an award for his pioneering work in Ecosystem Management. I remember his acceptance speech saying that the Chief should get an award for having the courage to give him an award for his controversial work.

New Perspectives

The next stage was something called New Perspectives. By now, problems managing the national forests were multiplying and the controversy growing more intense. Congress also was feeling the heat and wanted assurance that the Forest Service was getting on top of its land management problems. One of the congressional committees decided to hold an oversight hearing and the essence of the subject was, "Chief, what are you doing about all of these problems in the Forest Service?"

The usual procedure in getting prepared for a congressional hearing is for your staff to prepare draft testimony along with a 3- to 4-inch-thick briefing book. The staff meets with the Chief a day or two before the hearing to go over the proposed testimony and make sure the Chief is comfortable with the Forest Service statement.

Well, that was the first time I ever laid eyes on the words New Perspectives. The words jumped out at me and I remember asking a lot of questions like, "Why call it New Perspectives?" The staff explained that we could not use the term new forestry because of its controversial history and baggage. They convinced me that New Perspectives was a good, neutral term and it would be hard for anyone to disagree with. Besides, it pretty well represented our broadening concept of national forest management.

Once I got over the idea of this new terminology, I did some editing to make it a stronger statement about actions the Forest Service would take to get on top of our land management problems. New Perspectives was to be a creative, experimental program to pilot test different concepts of land management. Researchers and land managers were to work together as a team in addressing the issues on each national forest. It meant that we had to shift the priorities of our researchers, to which Jerry Sesco, the Deputy Chief for Research, agreed in advance. I was to direct each Forest Supervisor, through the Regional Foresters, to get together with the appropriate research scientists and design some pilot tests or experimental projects to demonstrate a new and more environmentally sensitive way of managing each national forest.

The briefing book included write-ups on all of the current and planned projects going on throughout the Forest Service that could be included in the New Perspectives program. Congressional hearings are a lot like final exams in college, except they are oral exams with Members of Congress asking the questions. Also, Members often have their own agendas, and they try to get the Chief to answer in a way that supports their individual viewpoints. So, after reading and studying all of the write-ups in my briefing book, I probably knew as much or more than anyone else in the Forest Service, on that particular day, about the intent and meaning of New Perspectives. It is almost as dangerous for the Chief to go to Congress to testify as it is to go on a walk in the woods—you never know what sort of policy might get made in the process.

So off to the Hill I went announcing the New Perspectives program as the new Forest Service way of dealing with many of our land management problems. As I recall, it was a friendly hearing with a lot of good questions by members. I am sure that over the course of the hearing, I must have added some details about the intent and meaning of New Perspectives. I think the Committee was generally convinced that the Forest Service was making a sincere and strong effort to get on top of our land management problems.

I was really proud of how quickly Forest Service field people responded in implementing New Perspectives projects. Before long, people could see on-the-ground results, including here in Arkansas on the Ouachita NF and the Ozark-St. Francis NF. It also set in motion an unprecedented degree of cooperation and teamwork among national forest land managers and researchers.

Ouachita NF Situation—The Walk In The Woods

In many cases, the Chief gets involved in national forest issues in which he does not have first-hand knowledge and experience, and has to depend on what other people tell him about the issues. However, in the case of the Ouachita NF, I did have some previous experience, even though it had been a long time ago. I graduated from the Forestry School at Arkansas A&M, now the University of Arkansas at Monticello, and learned a lot about southern pine management, including the uneven-aged management practices on the Crossett Experimental Forest. During my college years, I worked for two summers in the late 1950's on the Womble Ranger District (RD) at Mount Ida, Arkansas, mainly on a timber marking crew and a timber stand improvement crew. So, I had walked up and down a lot of mountains on the Ouachita NF and knew what the country and forest looked like before the Forest Service began large-scale clearcutting.

In the mid-1960s, I was the District Ranger on the Choctaw RD of the Ouachita NF. Ironically, I am the only Chief out of the 14 in the history of the Forest Service who has ever been a District Ranger. So, the Ouachita NF has the honor of being the only national forest ever to have produced a Chief out of its Ranger ranks.

I also remember participating in a Ouachita NF Ranger/Staff meeting here in Hot Springs in about 1966. We had been in the clearcutting business for about 2 years and we Rangers were getting some flak from the natives. At the Ranger/Staff meeting we expressed our concerns about clearcutting. As I remember, the Forest Supervisor and Timber staff dealt with our concerns rather directly and bluntly and told us our job was to go home and educate the public about the merits of clearcutting. So, I spent the next 2 years trying to do just that here on the Ouachita NF.

Well, I left the Ouachita NF in 1968 and did not get involved again until the famous walk in the woods in 1990, some 22 years later. So, I will tell you about that.

U.S. Senator, David Pryor of Arkansas was on our back about clearcutting on the Ouachita NF. He had written me a couple of letters asking me to look into the situation and see if we could stop it. My very competent staff in Washington, D.C., with the help of Region 8 staff in Atlanta, had prepared responses to Senator Pryor that should have convinced any reasonable person that clearcutting was an essential silvicultural tool for managing the Ouachita NF. However, Senator Pryor wasn't buying it. He wrote me another letter asking me to stop clearcutting on the Ouachita NF. Again, my staff prepared a response that told him the same thing, as though he did not quite understand the first letter. This time, however, my staff decided that the issue with Senator Pryor had escalated to the point that the Chief should sign the letter.

Well, when the letter arrived on my desk for signature and I reviewed its long history, I decided to rewrite it myself. As I recall, I wrote a short response saying that I, too, had some concerns about clearcutting in Arkansas and that we were experimenting with something called New Perspectives aimed at gradually getting out of the clearcutting business as a standard timber harvest practice. I then invited Senator Pryor to join me in a visit to the Ouachita NF to see first-hand how we were managing the Forest. He accepted my offer and we were on our way to Arkansas as soon as schedules could be worked out.

The walk in the woods, as it was later called, took place in August 1990 on the Winona Ranger District of the Ouachita NF. Along with Senator Pryor and myself were Regional Forester Jack Alcock, Forest Supervisor Mike Curran, Dr. Jim Baker from the Southern Forest Experiment Station, and District Ranger Norman Alley. Thus, we had line officers from the four levels of the Forest Service, plus a researcher—a potentially dangerous group, as the Rangers later found out! We spent the morning looking at some clearcuts, as well as some New Perspectives timber sales.

As a group, I think we were all impressed with the New Perspectives projects. I remember Jim Baker explaining to Senator Pryor how important it was to manage the sunlight so as to be able to regenerate the forest to desirable species. This really made an impression on the Senator. Also, I noticed that Ranger Alley had a twinkle in his eye and sense of excitement about his New Perspectives projects, which impressed me. And we never succeeded in convincing the Senator that clearcutting was an essential silvicultural tool on the Ouachita NF, especially since we had good examples of successful partial cutting methods under our New Perspectives program. We ended up at Lake Sylvia Recreation Area for a picnic lunch, which concluded our trip with Senator Pryor. The Ranger's wife got involved in providing us a fancy tablecloth and some really great food for lunch, which also really impressed the Senator.

Over lunch, the inevitable question came up, “Why can’t the Forest Service manage the entire Ouachita NF using the partial cutting practices that we had just seen on the New Perspectives projects and do away with clearcutting?” The main arguments against doing away with clearcutting at that time were:

(1) New Perspectives was a new, experimental program in its early stages of implementation and had not been proven successful yet, except for the fact that we could harvest timber in a way that was not an eyesore to the public.

(2) The jump from a few experimental New Perspectives timber sales to managing the entire Ouachita NF based on the principles of New Perspectives without the use of clearcutting had huge operational problems. How would you like to have been a Ranger at that time with the Chief and Regional Forester coming to town and turning your whole timber sale program upside down? It would be like trying to assemble a bicycle and ride it at the same time.

In spite of these good reasons to continue clearcutting on the Ouachita, Jack, Mike, and I knew down deep that we were on an unpopular, losing path and that some rather drastic changes were needed. We also knew that those changes would be difficult regardless of the timing. Dr. Jim Baker participated in the discussion and said that he was willing to shift his priorities to help the Rangers. Knowing that my old college classmate was willing to step up to the plate and put his professional reputation on the line gave me added confidence. Forest Supervisor Curran and Ranger Alley said they were willing to go for the change. We four line officers agreed that such a change would severely disrupt the timber sale program. However, we were all in this thing together and would take collective responsibility for any downfall in target accomplishments. The Rangers were not to be left hanging out on a limb alone.

So, we made the decision to designate the entire Ouachita NF as a New Perspective Forest, and the rest is history. That is why you are having this symposium this week. I hope you conclude that it turned out well!

This was not only a big decision for the Ouachita NF, but also a big one for the entire Forest Service. It caught everyone’s attention both inside and outside the agency. They suddenly understood the importance and implications of the New Perspectives program—that it was for real and that the Forest Service was getting serious about making a major conceptual change in how we managed the national forests.

Ecosystem Management

As time marched on, the New Perspectives program gained momentum, and more and more projects came on-line with good success. People began to describe what they were doing under the New Perspectives and Ecosystem Management programs. These terms began to find their way into the Forest Service vocabulary with some regularity. There was a fast-developing consensus in the agency that Ecosystem Management was the new concept that we had been searching for to guide future management of the national forests.

However, making a major policy change in the Washington, D.C., political environment is not easy and takes time. This is especially the case if it becomes controversial and could result in changing the way national forests are managed. So, I will tell you the story of how the policy change was made in Washington, D.C. Again, our old friend, clearcutting, played a crucial role, just as it did here in Arkansas.

Bill Riley was Administrator of the Environmental Protection Agency (EPA) during the Bush Administration. I knew Bill before he was appointed to the EPA job, and we would occasionally get together for lunch just to talk things over. Bill was concerned about clearcutting and the old-growth forests of the Pacific Northwest. We always ended up talking about these issues and Bill became quite knowledgeable about them.

Then in 1992, the Earth Summit (United Nations Conference on Environment and Development) was held in Rio de Janeiro, Brazil. Bill Riley was appointed head of the U.S. delegation. When he got to Rio, several representatives of American environmental groups and some Members of Congress were there questioning forestry practices in the United States and holding press conferences telling the world about how the U.S. was managing its forests. The main issues were clearcutting and old-growth forests—the very issues Bill was concerned about and had been talking to me about.

Bill and the other U.S. delegates were getting beat up pretty badly in Rio de Janeiro. I am sure that the other leaders from around the world were pretty confused by having an official delegation from the U.S. saying that we were doing a good job of managing our forests and an unofficial group of Americans, including some Members of Congress, vehemently disagreeing. President Bush was scheduled to go to Brazil and make a speech on the closing day of the Earth Summit.

The end of the Bush Administration was near, and there had been some major personnel changes in the White House. As it turned out, my old boss and friend, former Secretary of Agriculture Clayton Yeutter, was serving as the President’s Chief of Staff. During all of this controversy, Bill Riley was talking to Clayton in the White House. Because the U.S. was getting roughed up pretty badly, there was even discussion about whether the President should go to Brazil and subject himself to all the controversy. Bill asked Clayton to talk to me to see if the U.S. could make a policy statement about clearcutting on Federal lands and let President Bush announce the policy change in his speech at the Earth Summit.

Clayton called me from the White House. Falling back on my experience here on the Ouachita NF, I told him that the Forest Service was willing to eliminate clearcutting as a standard method of timber harvest on the national forests. However, we still needed to keep the option open for exceptional cases.

Then I made the big move! I called Clayton and told him I would also like to announce that we were officially adopting an Ecosystem Management policy for the national forests. There was a little pause and then he asked, “What in the world is Ecosystem Management?” I had about 5 minutes to explain it to him and describe why it was a good policy for the country. After he had heard my explanation, Clayton said that the policy sounded pretty good to him and asked that I write it up and fax it to him so he could discuss it with the President.

I went home that night and worked until about 11 p.m. on the proposed Clearcutting and Ecosystem Management policy statement. I got up at 3 a.m. the next morning and worked another 3 hours on it. I had to come up with wording that made sense to Forest Service employees, as well as to Clayton and President Bush in his speech in Brazil. At about 8 a.m., I faxed the statement to the White House. In about an hour, Clayton called me and said the President liked it and wanted to announce the policy in his speech at the Earth Summit.

In the meantime, back home at the Department of Agriculture, all was not well. My relatively new boss, Secretary of Agriculture Madigan, had not been in the loop and was unaware of what was going on. He was not an easy person to work with, and our relationship was a rather rocky one. Secretary Madigan, having no experience in forestry, thought that a good Chief would run a peaceful Forest Service without much controversy. By his standards, I was not performing very well. I knew that it would not be easy to get his approval. The Acting Assistant Secretary of Agriculture was John Beuter, a professional forester, and he was a big help in getting the policy through the Department. After we had met with Secretary Madigan’s key staff, they reluctantly agreed to go along with the new policy, especially since President Bush had already approved it. However, this situation certainly did not improve my rather poor standing with Secretary Madigan.

As it played out, I issued the famous policy letter of June 4, 1992, on clearcutting and ecosystem management as the President was leaving for Rio de Janeiro. The President gave his speech at the Earth Summit, and an important part of it was the new policy statement. So, that is the real story as to how it all happened in Washington, D.C.

CONCLUSIONS

I would just like to conclude by saying that the Chief has to stand on the shoulders of Forest Service field people. By being creative and working hard to make the new perspectives program successful, they were the real key players. They forged the thinking that became the Ecosystem Management concept. As Chief, I was able to marshal a unique fleeting opportunity to translate this thinking into a major policy change for future management of the national forests. As you can see, the Ouachita NF played a crucial role in how it all unfolded. And I thank you for that!

Well, all of you know the rest of the story. We had a presidential election in November 1992 and your guy from Arkansas won. The new Administration strongly supports ecosystem management, and has made it the centerpiece of their policy in managing the national forests. However, they wanted a Chief of their own choosing.

So, I rode off into the sunset to Arizona where I am pursuing a new life unencumbered by the forestry issues of the day. It has been my great pleasure to return to Arkansas and, after 9 years, to see strong evidence that the decisions I made as Chief are working out better than I had hoped.

OVERVIEW OF ECOSYSTEM MANAGEMENT RESEARCH IN THE OUACHITA AND OZARK MOUNTAINS: PHASES I–III

James M. Guldin[1]

Abstract—When the shift away from clearcutting and planting on the Ouachita National Forest was implemented in the early 1990s, it became apparent that research support for reproduction cutting methods that employ natural regeneration in shortleaf pine stands in the Interior Highlands was lacking. To fill that need, research scientists and land managers established a three-phase research program that included demonstration case studies, replicated stand-level studies, and landscape studies at the watershed scale. This review of each of these three phases includes a brief history and justification, an overview of objectives and a summary of the study design, and a report on the status and future work planned in each phase.

INTRODUCTION

The development of the Land and Resource Management Plan on the Ouachita National Forest (NF) in the 1980s triggered considerable debate. In 1986, the release of the first draft of the plan to the public called for the continued and widespread use of clearcutting and planting as a primary means to regenerate shortleaf pine (*Pinus echinata* Mill.) and pine-hardwood stands in Ouachita Mountains. That draft attracted 11 appeals from individuals and organizations, and was in part responsible for a change in leadership on the Ouachita NF (Curran 1994). Between 1986 and 1990, the Forest Supervisor on the Ouachita NF led the effort to revise the draft. With the release of the Amended Land and Resource Management Plan (U.S. Department of Agriculture, Forest Service 1990), all appeals except one were dropped (Curran 1994). The locally renowned walk in the woods by two native Arkansans, Forest Service Chief Dale Robertson and Senator David Pryor, is described elsewhere (Robertson, in press).

That walk in the woods led to the establishment of the Ouachita NF as a “Lead Forest” under the New Perspectives Program. The Southern Forest Experiment Station (now part of the Southern Research Station) was directed by the Chief to provide scientific support for a shift in management philosophy away from clearcutting and planting, and toward even-aged and uneven-aged high-forest reproduction cutting methods that rely on natural regeneration. The Monticello-Crossett Forestry Sciences Laboratory was headed at that time by Dr. James Baker, who had been a college classmate of Chief Robertson. That laboratory had considerable expertise in silvicultural treatments using natural regeneration of mixed loblolly (*P. taeda* L.)-shortleaf pine stands of the upper west Gulf Coastal Plain. However, there was not much experience there or elsewhere in applying even-aged or uneven-aged silvicultural systems using natural regeneration of the pure shortleaf pine stands found across the Interior Highlands (Baker 1994).

As a result, a research team was assembled to develop silvicultural options for shortleaf pine and pine-hardwood forests, and to study the effects of those options on a host of resource attributes and values of interest to managers and scientists. With the advent of the ecosystem management approach for national forests and grasslands in 1992 (Robertson, in press), the work of this team came to be known as the Ouachita Mountains Ecosystem Management Research Project. With the Southern Research Station in the coordinating role, 10 research units have played a part in the research program. Other cooperators have included the Ouachita NF, the Ozark-St.Francis NF, Region 8 headquarters of the USDA Forest Service, more than a dozen universities in the region, Weyerhaeuser Company, the National Council of the Pulp and Paper Industry for Air and Stream Improvement (NCASI), and several State agencies and nongovernmental organizations.

Highlights of the establishment of the program included close working support with academia, establishment of a research liaison position on the staff of the Ouachita NF to work with the research team, and the development of independent funding through supplemental appropriations from Congress and through NCASI. Since then, the working relationship between the Ouachita NF and the Southern Research Station has been exceptionally close.

ORGANIZATION OF THE RESEARCH PROGRAM

The shortleaf pine forest type covers a majority of the area managed for timber production on the Ouachita NF. This forest type is dominated by shortleaf pine and also contains a minor and varying hardwood component that includes white and red oaks (*Quercus* spp.), hickories (*Carya* spp.), and elms (*Ulmus* spp.). Shortleaf pine and pine-hardwood stands in this forest type are typically found on south and southwest-facing slopes in the Ouachita Mountains, the Arkansas River Valley, and the Boston Mountains in Arkansas and Oklahoma. As a result, research study sites were located on the Ouachita NF in Arkansas and Oklahoma as well as the southern portion of the main unit of the Ozark NF in the Boston Mountains of Arkansas.

The Chief’s interest was in the demonstration of alternatives to clearcutting and planting that could be quickly implemented and shown to professional resource managers and the

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public. That mandate quickly developed into a three-phase program of research and demonstration. Mersmann and others (1994) reviewed the origins of the project in detail, and they described operational planning procedures and public involvement in phases I and II.

Phase I—Demonstration Stands

The phase I demonstration stands were established in 1990–91 immediately after inception of the project. These demonstrations were intended as examples of the alternatives to clearcutting that were being considered. They allowed people inside and outside the agency to see and discuss those alternatives. They were established by modifying existing open timber sale projects; with contractor approval, sales were modified to illustrate the seed tree, shelterwood, single-tree selection, and group-selection reproduction cutting methods. Twenty-two demonstrations were established in three general areas: (1) the Winona Ranger District (RD) for tours originating in Little Rock, (2) the Womble RD for tours emanating from the Supervisor's Office in Hot Springs, and (3) the Kiamichi RD and Choctaw RD in eastern Oklahoma for tours originating there. The Womble RD tour stops were the most frequently visited. One of the phase I stands on the Womble RD that features classic uneven-aged foliar canopy structure in shortleaf pine is still used for demonstration.

By the time the phase I tours were generally retired in 1994, they had supported more than 50 tours, with an estimated attendance of more than 1,000 participants. As unreplicated case studies, these demonstration stands provided little of scientific value. But they had tremendous value as a tool for allowing people to envision the changes embodied in the new perspectives and ecosystem management concepts being put forth at the time.

Phase II—Stand-Level Research

The phase II study was designed to test different reproduction cutting methods using a replicated experimental approach, which provides a higher standard of statistical rigor than was found in the unreplicated phase I demonstrations. The two main objectives of the phase II study are (1) to evaluate biological and economic feasibility of even-aged and uneven-aged high-forest reproduction cutting methods that rely on natural regeneration to establish and maintain shortleaf pine and pine-hardwood stands in the Interior Highlands, and (2) to quantify the effects of these alternatives on a spectrum of forest resources and values.

The phase II study design called for a test of 13 reproduction cutting methods (table 1) in 4 ecoregional blocks for a total of 52 stands (Baker 1994, Guldin and others 1994). The 52 stands were randomly selected, and treatments were randomly assigned such that each stand would have 1 reproduction cutting treatment imposed upon it with 4 stands of each given treatment in the study. The random assignment of treatments led to an interesting effect. Because each stand in the study was eligible for any of the reproduction cutting methods that were being tested, and because the management plans for the national forests restricted the use of clearcutting in stands near roads, the phase II stands are all some distance from major roads in the region.

The phase II study went through the standard operating procedure for compliance with provisions for public involvement. Mersmann and others (1994) describe in detail the preparation of the environmental assessment for the phase II study. As a result, the entire phase II study was considered a single decision, and it was not appealed.

Research crews inventoried stands and prepared marking guidelines for the treatments being imposed in each stand. Those marking guidelines were given to the RD marking crews that marked the stands in an operational manner as part of the district workload. The research harvest generated

Table 1—Reproduction cutting methods and residual basal area for the pine and hardwood component within each in the phase II stand-level studies

	Residual basal area		
Reproduction cutting method	S/W/X	Pine	Hardwood
		square feet per acre	
Clearcutting	W	0	2 – 5
Seed tree method, pine	S	20	2 – 5
Seed tree method, pine-hardwood	S	10	10
Shelterwood method, pine	S	40	2 – 5
Shelterwood method, pine-hardwood	S	30	10
Shelterwood method, pine-hardwood	W	30	10
Group selection method, pine	X	60	2 – 5[2]
Group selection method, pine-hardwood	W	50	10[2]
Single-tree selection method, pine	S	60	2 – 5
Single-tree selection method, pine-hardwood	S	50	10
Single-tree selection method, pine-hardwood	W	50	10
Single-tree selection method, low-impact	X	60	15
Unmanaged control	X	~100	~30

S = included in split-plot site preparation and release study; W = included in wildlife group habitat study; X = not included in either the split-plot or wildlife studies.

10 million board feet of sawtimber—roughly 8 percent of the annual timber harvest on the Ouachita NF in 1993.

As an element of operational work, RD personnel prepared sale area improvement plans for harvested areas on their respective districts, and collected Knutsen-Vandenberg (KV) funds for planting, site preparation, release, and monitoring. Through this process, the Ouachita and Ozark-St. Francis NFs allocated roughly $1 million in KV funds to the Southern Research Station from FY94 through FY98 to monitor the effects of KV-funded activities in the 48 harvested stands. Monitoring was structured within a statistically rigorous study design and it involved repeated visits to stands.

Seven groups contributed to the research and monitoring activities in phase II stands:

1. The woody vegetation group studied seed production, seedbed condition, woody plant regeneration development, shortleaf pine genetic diversity, overstory development, and the effects of site preparation and release
2. The understory vegetation group measured density, frequency, and cover of herbaceous and shrub species on plots nested within the woody vegetation plots
3. The wildlife research group studied small mammals, flying squirrels, neotropical migratory and resident birds, and general wildlife habitat conditions
4. The management economics group quantified harvesting costs and management costs associated with various reproduction cutting methods
5. The visual quality group measured visual impacts of recently harvested stands, conducted customer surveys of scenic preferences, and evaluated the effects of hardwood retention, season, and physiography on perceived scenic beauty
6. The arthropod and microbial communities group studied insect diversity (with emphasis on arthropods), cone and seed insects, southern pine beetle hazard ratings, and crown health of hardwoods
7. The water, soil, and cultural resources group studied the water chemistry of ephemeral streams, herbicide movement in streamwater, stream channel morphology and woody debris, soil disturbance associated with logging, and harvesting effects on cultural resources.

Some of the results of these many studies are contained in this proceedings. Other papers can be found in the refereed literature.

Phase III—Landscape-Scale Research

Early in phase II design, it became apparent that some questions could not be answered at the stand level. These questions include effects of management on hydrology and aquatic ecology, and the ecological ramifications of maintaining or imposing different vegetation patterns across a watershed or landscape.

The phase III study was developed to support operational implementation of ecosystem management at the landscape scale, defined for these purposes as watersheds. Emphasis was placed on research on watershed hydrology and aquatic ecology, linkages between terrestrial and aquatic systems, landscape analysis of forest patterns and processes, landscape-level terrestrial wildlife concerns, and the social context of ecosystem management.

In the planning for phase III, scientists took advantage of smaller stand-level studies when that was possible. Two examples are studies of the use of prescribed fire and studies of the effects of retaining trees within group openings in stands being managed using the group selection method.

Four watersheds make up the overall study design of phase III research. Each watershed has a unique desired forest condition; taken together, the watersheds reflect a range of models for ecosystem management. Since replication at the scale associated with this phase is difficult, a different approach to experimental design was required. That approach builds on a combination of approaches, including the use of repeated measurements across years, subdivision of existing watersheds or establishment of new watersheds for validation of modes, and traditional small-scale replicated research within the larger watersheds (Guldin, in press).

Of the four watersheds that are included in the landscape-scale study, three are in the Upper Lake Winona Basin of the Winona RD on the Ouachita NF; the fourth is on nearby forest industry land (table 2). Management intensity varies widely. The Alum Creek watershed represents essentially an unmanaged condition. The Little Glazypeau watershed supports intensive management of pine plantations for industrial timber production. The other two watersheds—Bread Creek and North Alum Creek—differ by percentage of industry ownership and, thus, by intensity of management overall. Overall, the intensity of management established for the watersheds during the baseline pretreatment measurement period reflects that distribution of ownership.

Table 2—Watersheds included in the phase III study with subjective classifications of ownership and management intensity

Watershed	Area	Ownership		Management intensity
		N.F.	Industry	
	acres	- - - *percent* - - -		
Alum Creek	3,700	100	0	Unmanaged
Bread Creek	3,800	100	0	Low
North Alum Creek	9,800	50	50	Moderate
Little Glazypeau Creek	5,600	5	95	High

N.F. = national forests.

Specific objectives in the phase III study are

1. to quantify core watershed hydrology through a series of flumes and uncontrolled cross-section gauging stations, which will be used to model hydrological factors and cumulative hydrology effects up to basin scale
2. to characterize and quantify sensitive and critical elements of aquatic and riparian ecology in concert with the studies of watershed hydrology
3. to quantify terrestrial ecological relationships of vegetation pattern, ecological classification, wildlife, and biodiversity across the core watersheds
4. to characterize the social dimensions of the landscape in which the core watersheds lie, including study of the prehistoric, historic, and current relationships of people with the land.

Baseline measurements in the phase III study were initiated in 1994, and carried forth through 1998. Papers included in this proceedings present baseline measurements of various types for that pretreatment period.

Treatments to carry the watersheds from their existing condition to their desired future condition are underway. The desired future condition in the phase III watersheds represents a synthesis of results to date from the stand-level research study with the interests of land managers with both the national forest and forest industry. The watershed under intensive management will remain so. It will quantify the effects of continued plantation management over time at the large scale. Half of the watershed in an unmanaged condition will continue in the unmanaged condition, and the remainder will be subject to single-tree selection silviculture, generally thought to be the least intensive of the reproduction cutting methods in the arsenal of the silviculturist. The Bread Creek watershed, wholly under Forest Service ownership and managed using typical national forest treatments in the past, will continue to support standard agency actions. The North Alum Creek watershed will be split three ways to support two variations of group selection treatments and a shortleaf pine-bluestem habitat restoration treatment (Guldin, in press).

As in phase II, these phase III treatments are being imposed in an operational manner by the regular management staff of the cooperating forest management organization. On industry land, typical schedules of harvest and associated cultural activities associated with intensive management of pine plantations for commercial timber production will continue as they would normally occur. On the national forest watersheds, actions have been planned and are being executed by the staff and field crews of the Jessieville and Winona RDs. An administrative study plan was written and approved that contains provisions for monitoring supported by KV funds. Those monitoring data and other data collected on public and private land will be indispensable in helping scientists and forest land managers understand the effects of concentrating management activities in a forested watershed.

SUMMARY

This symposium is linked to larger regional science issues through the Southern Research Station's strategic science framework. A crosscutting theme in the Station's strategic plan highlights the sustainability and productivity of the Interior Highlands ecosystem. The Ouachita Mountains Ecosystem Management Research Program is a key element of that theme, and work reported in this proceedings supports it directly.

The three-phase approach of demonstration, stand-level research, and landscape-scale research demonstrates how team-oriented science efforts bring "critical mass" to complex experiments at heterogeneous scales. However, the research project would not be possible without many Station partners in public and private sectors who have contributed time, talent, in-kind services, and both direct and indirect financial support.

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Session 1
Phase II Silviculture Research

James B. Baker, Moderator

REGENERATION DEVELOPMENT ACROSS A RANGE OF REPRODUCTION CUTTING METHODS IN SHORTLEAF PINE AND PINE-HARDWOOD STANDS IN THE INTERIOR HIGHLANDS

James M. Guldin, James B. Baker, and Michael G. Shelton[1]

Abstract—Density, milacre stocking, and height of shortleaf pine (*Pinus echinata* Mill.) regeneration under 13 reproduction cutting methods were measured after 5 growing seasons across a range of reproduction cutting treatments in shortleaf pine and pine-hardwood stands in the Interior Highlands of Arkansas and Oklahoma. A subset of the full database was used to suggest trends in the data to date. After five growing seasons, six of the treatments exceeded the minimum standards for regeneration success—the clearcutting treatment, two of the three shelterwood treatments, and three of the four single-tree selection treatments. The two seed-tree methods fell short in both pine regeneration density and pine milacre stocking, whereas the two group selection treatments and the unmanaged control fell short in both pine milacre stocking and height. Key questions over the next 5-year period are whether regeneration development can be maintained as residual overstory basal area continues to increase, and whether the 10-year cutting cycle harvest in the uneven-aged stands or the partial removal cut in the even-aged stands will cause unacceptable mortality in the current regeneration cohort.

INTRODUCTION

Shortleaf pine (*Pinus echinata* Mill.), the most widely distributed of the four major southern pines, reaches its ecological optimum in the Ouachita Mountains. In this region, the topography is characterized by long ridges running generally from east to west. South-facing slopes are dominated by pine and pine-hardwood stands, and the pine component consists exclusively of shortleaf pine. No other part of the country contains naturally regenerated stands in which shortleaf pine is the major, dominant, and only pine species over so large an area of pine-dominated landscapes.

There has been relatively little interest in the silviculture of natural stands dominated by shortleaf pine. Little is known about regenerating shortleaf pine with even-aged and uneven-aged reproduction cutting methods. However, interest in natural regeneration of shortleaf pine has increased with the shift in management philosophy away from clearcutting and planting on the Ouachita National Forest in the 1990s.

In this paper, we describe results of a variety of even-aged and uneven-aged high-forest reproduction cutting methods to establish and maintain shortleaf pine and pine-hardwood stands in the Ouachita Mountains and southern Boston Mountains of Arkansas and Oklahoma. Key silvicultural questions are (1) the early growth of seed-origin shortleaf pine versus sprout-origin hardwoods, (2) the effects of overstory hardwoods on regeneration development, and (3) the effects of residual basal area on pine and hardwood regeneration development. We present initial data from a subset of treatments, and we give suggestions for further analysis.

METHODS

Study Layout

The study layout was established as previously described (Guldin and others 1994). Fifty-two stands were included in the study. In each stand, 12 plots were assigned to the timber management zone (TMZ) where tree harvesting was permitted and 2 plots were installed in streamside management zones, where harvesting is excluded. All plot centers were located using geographic positioning systems, and were digitized on low-level orthogonally rectified aerial photos.

Reproduction Cutting Treatments

Even-aged and uneven-aged reproduction cutting methods was tested. Even-aged methods were the seed tree and shelterwood methods. Uneven-aged methods included both the single-tree selection and group selection methods. At least two variations of each method were tested—one in which primarily pines were retained in the residual overstory, and one in which both pines and hardwoods were retained. In addition, two control treatments were used—the clearcutting method and an unmanaged treatment. Table 1 lists the residual basal area target for pines and hardwoods in each treatment, and the site preparation and release treatments used in each treatment.

Site Preparation and Release Treatments

A subset of stands was subdivided into quarters to measure effects of site preparation and release (Table 1). One quarter remained untreated by either site preparation or release. A second quarter received two manual felling treatments—manual site preparation early in the first growing season in the late spring and early summer of 1994, and manual release early in the fifth growing season in the late spring and early summer of 1998. Manual felling was done using either machetes or chain saws. In a third quarter, chemical site preparation was imposed by applying triclopyr (Garlon®) to cut surfaces of vegetation prior to the first growing season in the late spring of 1994. In the fourth quarter, chemical release was conducted by applying triclopyr (Garlon®) as a cut surface application or a directed foliar spray immediately prior to the fifth growing season in the late spring and summer of 1998.

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Table 1—Reproduction cutting method treatments included in the phase II study in the Interior Highlands

	Site preparation/ release treatments[a]				RBA	
Reproduction cutting methods	Q1	Q2	Q3	Q4	Conifer	Hardwood
					square feet per acre	
Clearcutting	C/N	C/N	C/N	C/N	0	2 – 5
Seed tree, pine	N/N	C/N	M/M	N/C	20	2 – 5
Seed tree, pine-hardwood	N/N	C/N	M/M	N/C	10	10
Shelterwood, pine	N/N	C/N	M/M	N/C	40	2 – 5
Shelterwood, pine-hardwood	N/N	C/N	M/M	N/C	30	10
Shelterwood, pine-hardwood	M/M	M/M	M/M	M/M	30	10
Group selection, pine	M/M	M/M	M/M	M/M	60	2 – 5
Group selection, pine-hardwood	M/M	M/M	M/M	M/M	50	10
Single-tree selection, pine	N/N	C/N	M/M	N/C	60	2 – 5
Single-tree selection, pine-hardwood	N/N	C/N	M/M	N/C	50	10
Single-tree selection, pine-hardwood	M/M	M/M	M/M	M/M	50	10
Single-tree selection, low impact	M/M	M/M	M/M	M/M	60	15
Unmanaged control	N/N	N/N	N/N	N/N	~100	~30

RBA = residual basal area; C = chemical; N = no treatment; M = manual felling.

[a] Site preparation/release treatment combinations are shown by quarter (Q1-Q4) to illustrate whether a given treatment was split or uniformly treated; split treatments were randomly assigned by quarter by stand.

Table 2—Measurements to date taken by the silviculture research group in the stand-level phase II study in the Interior Highlands

Schedule	Season	Data collected
Preharvest	Winter 1992–93	Overstory
Harvesting occurred	Summer 1993	None[a]
Postharvest year 0	Winter 1993–94	Overstory
Postharvest year 1	Winter 1994–95	Regeneration
Postharvest year 2	Winter 1995–96	Overstory
Postharvest year 3	Winter 1996–97	Regeneration
Postharvest year 4	Winter 1997–98	Overstory
Postharvest year 5	Winter 1998–99	Regeneration

[a] None taken during harvest.

In a second subset of treatments, competing vegetation in the entire stand was manually felled as described earlier (table 2). The purpose of this treatment was to promote uniform understory habitat conditions for studies of bird populations.

The group selection treatments and the controls were omitted from the split-plot site preparation and release comparison (table 1).

Measurements

Overstory vegetation (woody plants 3.6 inches in d.b.h. and larger) and regeneration (woody plants 3.5 inches in d.b.h. and smaller) were measured separately. Each measurement cycle required approximately 6 months of field work and was restricted to the dormant season, so overstory and regeneration measurements occurred on a 2-year cycle. Measurement cycles completed to date are shown in table 2. Measurement protocols for overstory and regeneration measurements have been published elsewhere (Guldin and others 1994) and are summarized later.

A nested sample was taken to measure the overstory. All woody plants 9.6 inches in d.b.h. and larger were sampled by species using variable-radius plots established using a prism with a basal area factor of 5.0 square feet per acre, and d.b.h. was recorded to the nearest 0.1 inch. After the harvest, all sample trees were tagged for remeasurement and the slope distance and slope percent from plot center to each tree were recorded. A measurement error of +5 feet was deliberately added to the formula used to calculate the horizontal limiting distance such that all trees up to and 5 feet past the exact limiting distance for the prism were sampled. The use of field data recorders and uploaded spreadsheet template data files allowed this calculation to be easily made in the field. In this way, borderline trees in the prism sample were included in or excluded from the sample by office computation rather than field technique.

All woody plants with d.b.h. between 3.6 and 9.5 inches inclusive were sampled by species on a 0.1-acre fixed-radius plot. The preharvest measurement was made by tallying these trees to the nearest 1-inch diameter class. After harvest, the d.b.h. of these was measured to the nearest 0.1-inch, and all sample trees were tagged.

The regeneration measurements were slightly modified from the original study design. Two separate samples—an inventory tally and a tagged tree tally—were taken. The inventory tally was designed to sample regeneration 1 year after the harvest, but was subsequently modified to sample the entire cohort of woody plants with d.b.h. ≤ 3.5 inches. Thus, in postharvest year 1, all woody plants between 3 inches in total height to > 4.5 feet in height but with a d.b.h. ≤ 0.5 inch were tallied on 6 milacres located within the 0.1-acre fixed radius plot. In postharvest year 3 and subsequently, all woody plants between 3 inches in height but with a d.b.h. ≤ 3.5 inches were tallied on the same 6 milacres.

The tagged tree measurement was designed to follow height and diameter growth of specific stems. Milacres were divided into four quadrats along cardinal directions. In postharvest year 1, the tagged tree sample was imposed on 2 milacres per plot; in postharvest year 3 and subsequently, the tagged tree tally was extended to all 6 milacres. During the first measurement, the tallest pine and the tallest hardwood, if present, per quadrat were tagged and measured for species identity, root-collar diameter, and total height. During remeasurement all previously tagged trees were remeasured; in addition, if a different pine or hardwood was taller than a previously tagged pine or hardwood, the new dominant was sampled and tagged.

ANALYSIS

A subset of stands based on rank within treatments was selected that would be representative of general trends in regeneration success from reproduction cutting methods. Results are summarized from only one of the four stands in each treatment. A broader analysis of the complete data set will be reported elsewhere.

The decision variable upon which the subset is based was milacre stocking after the fifth growing season. Thus, in all treatments except the clearcutting treatment, the four stands were ranked by percent milacre stocking in the pine component within the TMZ. The stand with the second-lowest milacre stocking was included in this subset to represent that treatment. The clearcut treatment was represented by the one stand in that treatment with the highest pine milacre stocking.

Milacre stocking was calculated as the proportion of the 72 milacres per stand stocked with at least one pine seedling. Regeneration density for both pine and desired hardwoods, defined as white oak (*Quercus alba* L.) and southern red oak (*Q. falcata* Michx.), was calculated as the sum of all pines or desired hardwoods on the 72 milacres per stand converted to a per-acre basis.

Pine height was calculated by averaging the height of all tagged pines after the fifth growing season. Thus, this variable represents the average height of the tallest pines in the regeneration cohort, not the average height of all pines in the regeneration cohort.

Acceptable standards for pine and hardwood regeneration density were set at 300 trees per acre in even-aged stands, and 200 trees per acre in uneven-aged stands (Baker and others 1996). These are the standards used by the Ouachita National Forest. If the trees are well spaced, the standards translate to 30 and 20 percent pine milacre stocking for even-aged stands and uneven-aged stands, respectively. Minimum acceptable height growth for pines is 0.5 feet per year; therefore, the third standard of acceptable height growth is whether seedlings exceed 2.5 feet in height after the fifth growing season.

RESULTS

Regeneration Density

In the even-aged treatments, acceptable regeneration density for shortleaf pine and for desired hardwoods is 300 trees

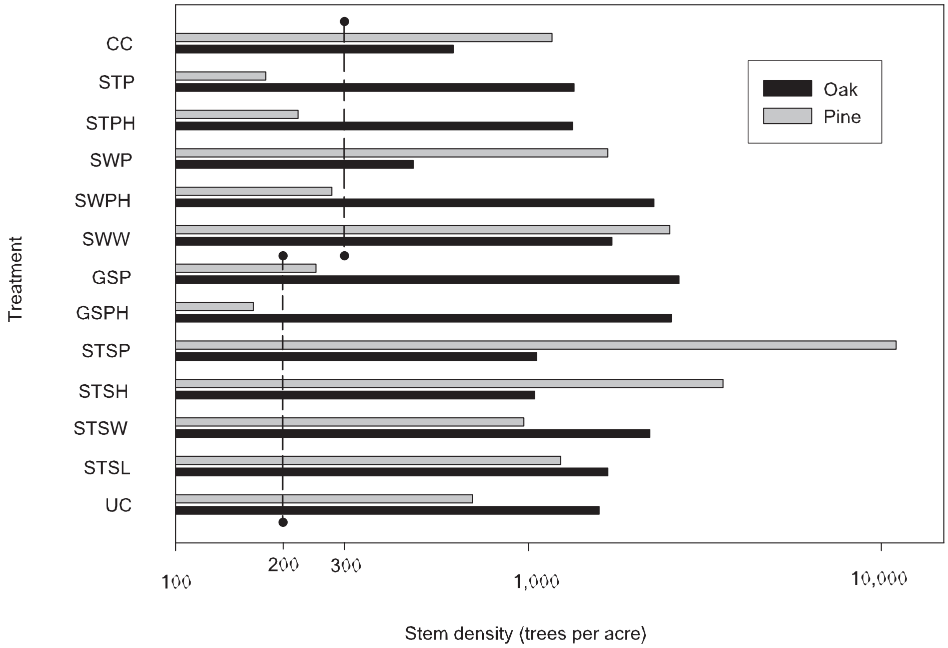

Figure 1—Density of regeneration, shortleaf pine, and combined white and southern red oaks after five growing seasons, by treatment (cf. table 1). Dashed vertical lines represent minimum acceptable standard for corresponding treatments.

per acre (fig. 1). The clearcut treatment supported more than three times this number of pines, and more than double the minimum number of desired hardwoods. The pine regeneration density was not surprising, because the clearcut was planted in the spring of 1994 with genetically improved 1-0 shortleaf pine seedlings on an 8-foot x 10-foot spacing (544 trees per acre). Scattered volunteer pine seedlings that seeded in naturally from adjacent stands boosted stocking. Virtually all of the white and southern red oaks are of stump sprout or seedling sprout origin.

Desired hardwood regeneration exceeded the minimum acceptable stocking in all even-aged seed tree and shelterwood treatments (fig. 2). Four treatments out of five had more than 1,000 stems per acre of desired hardwood regeneration.

In two of the three shelterwood treatments, pine regeneration exceeded the minimum acceptable density by between threefold and fourfold. Pine regeneration density was also greater here than in the clearcut stands. The third shelterwood treatment yielded just slightly less than the minimum acceptable density. Conversely, both seed-tree treatments fell well below the acceptable minimum density of pine regeneration.

The minimum acceptable regeneration density for shortleaf pine and for desired hardwoods in the uneven-aged treatments is 200 trees per acre. All uneven-aged treatments exceeded this minimum number in the desired hardwood component, varying between 1,000 and 2,000 stems per acre.

In the pine component, all of the single-tree selection treatments produced adequate pine regeneration density. Only one of the two group selection treatments yielded an acceptable number of pines. Stem density in the single-tree selection treatments varied from 800 to 10,000 trees per acre.

The unmanaged control treatment supports acceptable densities of pine and hardwood regeneration as well (fig. 1). More than 700 pines and 1,500 desired hardwood seedlings and saplings per acre are present.

Milacre Stocking

In the even-aged treatments, minimum acceptable milacre stocking of shortleaf pine is 30 percent. After the fifth growing season, milacre stocking of pine exceeded 60 percent in the clearcutting treatment (fig. 1). The unmanaged control, as expected, had the lowest pine milacre stocking of all treatments—< 10 percent.

In the even-aged seed tree and shelterwood stands, milacre stocking of shortleaf pine was much lower than in the clearcut stand. However, pine milacre stocking in two of the three shelterwood treatments exceeded the minimum of 30 percent, and the third was < 5 percent below the minimum. Both seed-tree stands had pine milacre stocking less than half the minimum acceptable stocking.

For the uneven-aged single tree and group selection treatments, minimum acceptable milacre stocking of pine is 20 percent (fig. 2). Three of the four single-tree selection treatments exceeded this minimum. The fourth, the low-impact single-tree selection, contained < 2 percent below the minimum. Neither of the group selection treatments contained more pine seedlings than the minimum.

Height of Shortleaf Pine Regeneration

In all treatments, acceptable development of dominant pines was judged by whether seedlings exceeded 2.5 feet in height after the fifth growing season. As with the other standards

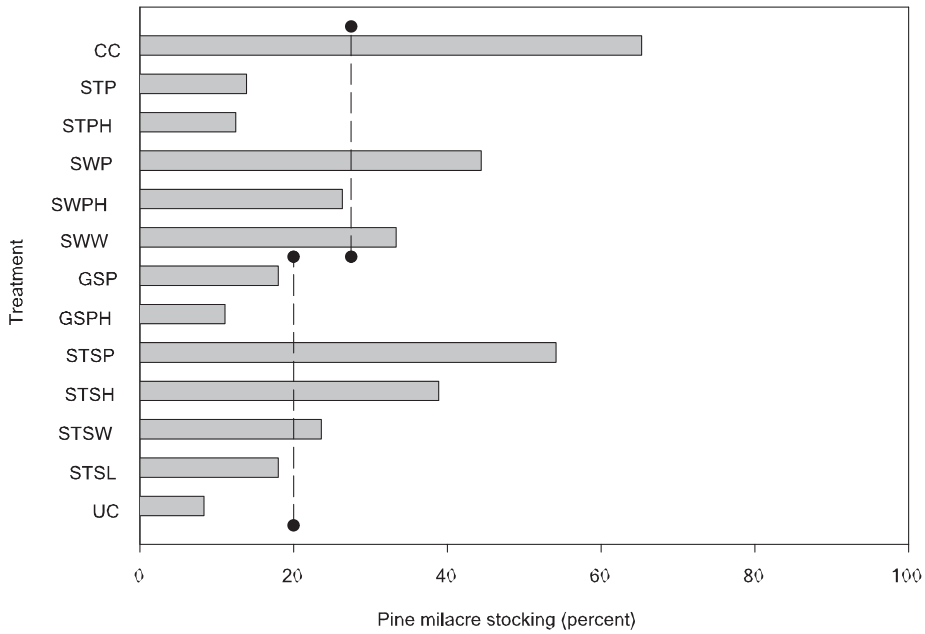

Figure 2—Milacre stocking, percent, for shortleaf pine regeneration by treatment (cf. table 1). Dashed vertical lines represent minimum acceptable standard for corresponding treatment.

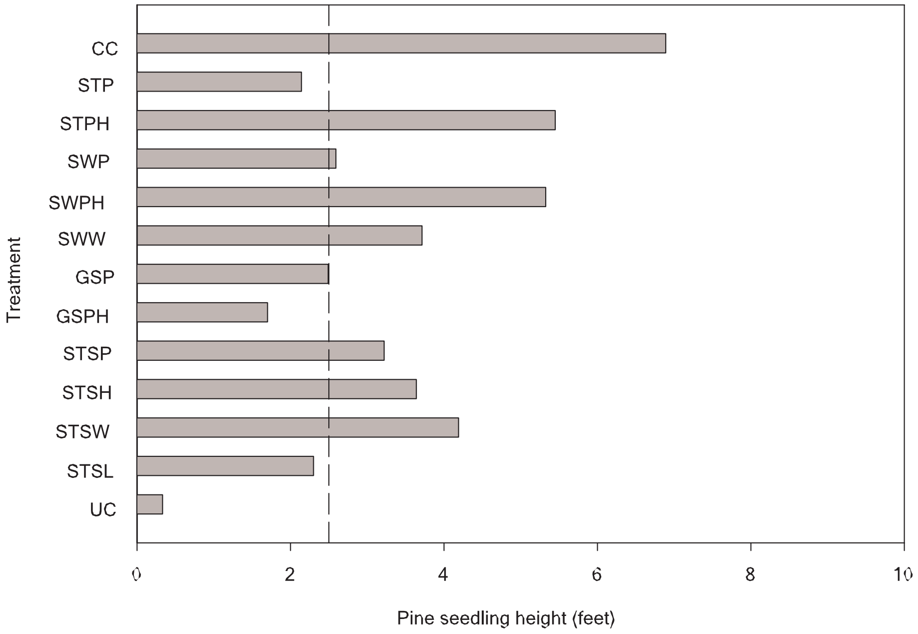

Figure 3—Height of dominant shortleaf pine saplings after five growing seasons by treatment (cf. table 1). Dotted vertical line represents minimum acceptable standard across all treatments.

of regeneration success, the clearcut treatment was best, with dominant pines exceeding 6 feet in height (fig. 3). Of the other five even-aged treatments, four exceeded the minimum standard, and two supported dominant pines whose average height exceeded 5 feet.

In the uneven-aged stands, the average height of dominant pines exceeded the minimum in three of the four single-tree selection stands, and was fractionally less than the minimum in one of the group selection stands (fig. 3). In all of the uneven-aged stands, the average height of dominant pines was < 5 feet after 5 years.

The average height of dominant pines in the unmanaged control stand was < 0.5 foot after five growing seasons. Seedlings in this treatment apparently can become established and can persist, but they grow very little.

DISCUSSION

Judging success of pine regeneration after five growing seasons requires consideration of three standards—acceptable density, acceptable stocking, and acceptable growth. As expected, the clearcut treatment has been most successful. Pine regeneration density exceeded 1,000 trees per acre, milacre stocking of 60 percent was double the minimum, and the height of dominant pines exceeded 6 feet after 5 years. These data all exceeded minimum standards by a comfortable margin.

However, several of the reproduction cutting treatments that rely on natural regeneration also exceeded all three standards. Specifically, two of the three shelterwood stands met or exceeded minimum standards; the third, which fell slightly below minimum in both regeneration density and milacre stocking, exceeded the other two in average dominant pine height.

Three of the four single-tree selection treatments also exceeded all three minimum standards, and the fourth, the low-impact single-tree selection stand, was only slightly below the minimum in both regeneration density and milacre stocking. The relatively high levels of overstory basal area in these single-tree selection stands was expected to hinder regeneration development, but this was apparently not the case over the first 5 years of the study. However, these stands are expected to grow approximately 2 square feet per acre of basal area annually. Thus, over the typical 10-year cutting cycle expected in these stands, the most critical years for overstory suppression of regeneration development are probably the second, rather than the first, 5-year period.

Between 1992 and 1996, about 60 percent of the even-aged reproduction cuts on the Ouachita National Forest were done using the seed-tree method; between 1992 and 1995, about one-fifth of the uneven-aged reproduction cuts on the same forest were done using group selection (Guldin and Loewenstein 1999). The data presented here suggest that these two treatments do not consistently exceeded minimum standards for regeneration establishment and development. Both seed-tree stands yielded less that the minimum standard in pine density and pine milacre stocking; both group selection treatments yielded less than the minimum standard in pine density and pine height growth.

A decision about whether a stand is successfully regenerated is commonly required on national forests during the first 5 years after harvest. These data raise a question about that standard, especially in uneven-aged stands. Overstory basal area is likely to reach a maximum during the last half of the cutting cycle rather than the first half. Seedling performance should be followed through the next cutting cycle harvest to determine whether regeneration development continues to be acceptable in the single-tree selection stands.

Before they can comfortably rely on natural regeneration, foresters need to know whether acceptable density and stocking of regeneration can be retained through the scheduled removal cut in the even-aged seed tree and shelterwood stands, and through the next cutting cycle in the uneven-aged stands. In this study, both are scheduled for the summer of 2003, the tenth growing season of this regeneration cohort. Ideally, this study should be continued beyond that point, so that these and other unresolved questions can be addressed.

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A COMPARISON OF VEGETATION WITHIN AND OUTSIDE RIPARIAN AREAS BORDERING EPHEMERAL STREAMS IN THE OUACHITA MOUNTAINS

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Abstract—Ephemeral streams frequently occur in shortleaf pine (*Pinus echinata* Mill.) hardwood stands that grow on the upper and mid-slopes of the Ouachita Mountains in Arkansas. Stream management zones are established around these ephemeral streams in the Ouachita National Forest to minimize impacts of adjacent forest management activities. To better understand the vegetation communities in these riparian areas, we quantified composition, density and diversity of the woody vegetation within these riparian areas and upland areas outside these zones. Overstory density outside riparian areas was significantly greater than inside ($p < 0.001$) but seedling density was significantly higher inside than outside the riparian area ($p = 0.07$). The overstory was more diverse within the riparian areas than outside. However, midstory, sapling, nor seedling diversity significantly differed between the two areas. Pines dominated the overstory in riparian and nonriparian areas alike. The midstory, sapling, and seedling size classes contained a greater species richness and were not dominated by any one single species as was the overstory.

INTRODUCTION

Riparian areas are located directly adjacent to water and are a transition between upland and aquatic ecosystems (Svejcar 1997; Hansen and Law 1994). These riparian areas have many essential functions. They display high levels of diversity (Naiman and others 1998), are important in maintaining water quality (Anderson 1992), reduce flood occurrence (Stuart and others 1994), provide shade for streams thereby reducing water temperature, and provide shelter for birds and other animals (Anderson 1992). The extent of a riparian area surrounding rivers varies depending on the type of river system, vegetation, topography and soils in the landscape.

Ephemeral streams primarily flow during snowmelts and rainstorms, thereby producing temporally and seasonally diverse environmental conditions for plant growth and adaptation (Banner and MacKenzie 1998). Given the spatially and temporally diverse nature of ephemeral streams, vegetation bordering these streams could be unique because of the need for plants to survive in both seasonally hydric and xeric environments. Species diversity in these areas could also be high because diversity and richness is frequently found to be greatest in areas where the disturbances are ephemeral in frequency and variable in their size and intensity (Malanson 1993). Thus, ephemeral riparian systems have the potential to be important for enhancing landscape and species diversity.

Within the Ouachita Mountains there are many ephemeral streams. These streams likely comprise greater than 50 percent of stream channels within this region (Personal communication. 1999. Alan Clingenpeel, Ouachita National Forest). Within the Ouachita National Forest, stream management zones (SMZ) or buffers in upland pine-hardwood stands are used to protect ephemeral streams and sustain the functions of the riparian vegetation and soils bordering these streams (U.S. Department of Agriculture, Forest Service 1990). Guidelines require that SMZ of at least 10 m width be installed on each side of ephemeral stream channels that are scoured (U.S. Department of Agriculture, Forest Service 1990). If a steep slope borders the stream, the SMZ is to be extended to the top or break of the slope. Forest management activities such as tree removal are minimized in SMZ to reduce disturbance to these areas. SMZ along ephemeral streams account for approximately 8-14 percent of area within shortleaf pine-hardwood stands growing on southwestern facing slopes in the Ouachita Mountains (David K. Radabaugh. 1999. Unpublished data. On file with: School of Forest Resources, University of Arkansas at Monticello, Monticello, AR 71656).

Although the primary objective of installing SMZ along ephemeral streams in the Ouachita Mountains is to protect water quality, land managers expect that these SMZ will also increase landscape diversity and wildlife habitat. However, to our knowledge there have been few if any studies in this region to quantify whether these ephemeral SMZ accomplish any of these secondary objectives. It is not known whether or to what degree the vegetation within these riparian communities differs from the surrounding upland forest communities. Without a fundamental knowledge of the vegetation bordering ephemeral streams it is difficult, if not impossible, to determine their value for fulfilling stand and landscape management objectives or developing management plans to sustain the ecological functions of these areas.

Characterization of the woody vegetation is the initial step to fully understand and better manage riparian communities that border ephemeral streams. Therefore, the objectives of this study were (1) to quantify the woody vegetation composition and examine the vegetative community characteristics, such as diversity and density, within SMZ adjacent to ephemeral streams in pine-hardwood stands and (2) to compare the composition and characteristics of plant

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communities within these SMZ to plant communities outside riparian areas in pine-hardwood stands located in the Ouachita Mountains.

METHODS

Study Design and Location

Woody vegetation measurements utilized for this study were collected at two different times. The first measurements were collected in 1993, from 16 relatively undisturbed, mature shortleaf pine-hardwood stands occurring on generally south-facing slopes in the Ouachita Mountains. These 16 stands were a subset of the stands used for the Phase II Ouachita/Ozark Ecosystem Management Study described by Baker (1994). Stands had a minimal size of 14 ha, were dominated by shortleaf pine that were at least 70 years old, occupied south or south-west slopes, and contained between 13.8 to 25.3 m^2 per ha of pine as well as between 4.6 to 11.5 m^2 per ha of hardwood basal area (Baker 1994). Overstory and midstory tree measurements utilized for our study were from two woody overstory and midstory plots established in the SMZ as well as from two of twelve overstory and midstory plots established outside the SMZ (Guldin and others 1994) in each of the stands. Plots outside the SMZ were selected such that they occurred on similar slope positions and as near as possible as those within the SMZ.

Saplings and seedlings were measured during 1999 in four of the original 16 stands. These four stands had been selected as controls, as part of the Phase II USDA Forest Service Ecosystem Management Study, and were the only stands of the original 16 that were not harvested. Three plots corresponding to lower-, mid-, and upper-slope positions were established in at least one SMZ in each of these four stands. The plots were 15 m wide while the length of the plot spanned the entire width of the SMZ. Thus plot length varied from location to location but averaged 32.6 m. The plots were placed at least 120 m apart and 75 m or more from the stand boundary or the upslope end of a SMZ. An additional 15 x 30 m plot was established adjacent to each of these plots but at a distance of at least 40 m outside the edge of the SMZ. The plots outside the SMZ were located at the same aspect and slope position to the corresponding plot located within the SMZ.

Overstory and Midstory Measurements

Trees with a diameter at breast height (d.b.h.) ≥ 24.4 cm were classified as overstory while trees with a d.b.h. ≥ 9.1 and < 24.4 cm were classified as midstory trees. Overstory trees were sampled using variable radius plots. Species and d.b.h. to the nearest 0.25 cm was recorded for each tree sampled. All midstory trees were tallied by 1 cm classes and species on a 0.04 ha fixed radius plot. A complete description of the midstory and overstory sampling and measurement methodologies can be found in Guldin and others (1994).

Sapling and Seedling Measurements

Woody stems > 3 cm in height with a d.b.h. ≤ 1.3 cm were categorized as seedlings. Seedling presence and coverage was quantified using 1 m^2 quadrats. A transect was established along the length of the plot such that it passed through the plot center. The quadrats were centered on the transect at each sampling location. Sampling locations were established at equal distances from each edge of the plot so that they mirrored each other along the length of the plot. The first quadrat in the SMZ plot was placed 2 m from the SMZ edge. The other quadrats were placed 5 m from the first. When the middle of the plot was less than 5 m away from a potential quadrat location, the quadrats were located half the distance from the last established quadrats and the center of the plot. Quadrats were evenly dispersed along the transect in plots located outside the SMZ. Density was determined by tallying the number of individuals of each specific species located within the quadrats.

Saplings were defined as woody species with a d.b.h. > 1.3 cm and dbh < 9.1 cm. Species and d.b.h. of each sapling in the 2 x 5 m plots that enclosed each of the 1 x 1 m seedling plot were recorded. D.b.h. was measured to the nearest 0.1 cm.

Data Analysis and Summaries

Density was determined for overstory, midstory, saplings and seedlings. Species diversity for each woody vegetation size class and plot was calculated using the Shannon-Weaver index (Shannon and Weaver 1949). Species richness values were also determined for each plot in each area. To determine the degree of similarity between riparian and nonriparian woody vegetation Sorensen index (Sorensen 1948) and Ellenberg index (Ellenberg 1956) were computed for each vegetation size class. Paired t-tests were used to test whether diversity, richness, and density significantly differed between nonriparian and riparian plots.

Importance values were used to evaluate the differences in species composition of the vegetation communities within and outside the riparian areas. Importance values of seedlings were calculated using relative density (stems per ha), relative dominance (percent cover) and relative frequency (individual species) (Krebs 1985). Importance values of the sapling, midstory and overstory were calculated using relative density (stems per ha), relative dominance (basal area per ha), and relative frequency (individual species) (Krebs 1985).

RESULTS AND DISCUSSION

Density

Overstory stems per ha and basal area per ha were significantly greater ($p < 0.01$) in the plots inside than outside the SMZ (table 1). On average the overstory in the nonriparian plots contained 34 percent more trees and 36 percent more basal area than overstory in the riparian plots (table 1). Differences in density of the overstory did not appear to be related to differences in growth or stand dynamics because average quadratic mean diameters did not differ between areas. It is more likely that the lower density in the riparian areas reflected a reduction in growing space due to the shallow, rocky soils on the steep slopes that often border these streams and area occuppied by the stream channels themselves. The shallow soil depths and rocky nature of the slopes bordering many of these streams would likely reduce tree survival and growth of trees in these areas. Crow and others (2000) indicated that reduced moisture holding capacity of sandy soils in riparian areas may cause trees to be scattered with low basal areas. Soils surrounding the ephemeral streams did not appear to be coarser than the areas

Table 1—Mean stem density (stems ha^{-1}), basal area (m^2 ha^{-1}) and probability level associated with paired T-test comparison of riparian and nonriparian plots

Size class	Riparian	Nonriparian	P-value
	Stem density		
Overstory	185	248	0.002
Midstory	491	480	0.790
Sapling	1516	1564	0.926
Seedling	6640	3733	0.072
	Basal area		
Overstory	16.7	22.3	< 0.001
Midstory	8.6	8.0	0.455
Saplings	2.0	2.6	0.498

outside the SMZ and thus the cause of the reduced density in the SMZ. Riparian plots generally had higher seedling densities than nonriparian plots. Differences were signficant at $p = 0.072$ (table 1). Midstory and sapling densities were similar in the two areas. The higher densities of the seedlings within the riparian plots may reflect the lower densities of the riparian overstory. The lower density of the overstory in the riparian plots most likely provides light conditions underneath the canopy that are more condusive to survival and regeneration of trees in the lower strata than does the higher overstory density in the nonriparian plots. On average, the riparian plots contained 78 percent more seedlings than the nonriparian plots.

Diversity and Similarity

The riparian overstory was significantly more diverse ($p < 0.01$) than the nonriparian overstory (table 2). Differences in diversity, in part, occurred because of the greater dominance of shortleaf pine in the nonriparian overstory. Pines composed 92 percent of all overstory stems in the nonriparian plots compared to 71 percent in the riparian plots. The higher diversity in the riparian plots also reflected greater species richness in these plots (table 2). Species richness of the overstory was significantly greater in riparian plots than in the nonriparian plots (table 2). Midstory, saplings and seedlings diversity nor species richness significantly differed between the two areas (table 2). However, richness and diversity values were consistently greater in the riparian areas regardless of size class.

Table 2—Mean Shannon-Weaver's index computed using stem density and basal area, mean richness, and probability level associated with paired T-test comparison for nonriparian and riparian plots

Size class	Riparian	Nonriparian	P-value
	SWI (Stem density)		
Overstory	0.65	0.27	< 0.001
Midstory	1.52	1.39	0.201
Sapling	1.21	1.10	0.645
Seedling	0.92	0.89	0.880
	SWI (basal area)		
Overstory	0.62	0.25	< 0.001
Midstory	1.46	1.31	0.157
Sapling	0.96	0.99	0.830
	Richness		
Overstory	4.25	2.69	0.006
Midstory	8.81	8.13	0.086
Sapling	4.25	4.00	0.377
Seedling	3.83	3.17	0.221

SWI = Shannon-Weaver's index.

It was apparent that diversity differed among size classes within these areas. Overstory and seedlings were the least diverse while the midstory size class was the most diverse. The large diversity values in the midstory in part reflects a greater diversity in the light environment of this strata. The midstory size class includes shade intolerant species as well as a number of mid-tolerant or shade tolerant species. The dominant overstory species such as shortleaf pine and post oak (*Quercus stellata* Wangenh.) are shade intolerant or in the case of white oak (*Q. alba* L.) mid-tolerant. Trees that dominate the midstory also included species such as blackgum (*Nyssa sylvatica* L.) and winged elm (*Ulmus alata* Michx.) that are shade tolerant. Sapling and seedling size classes are primarily dominated by shade tolerant species while importance values of shade intolerant species were much lower than in overstory or midstory size classes (tables 3-6). Differences in diversity among woody vegetation size classes may also reflect the number and distribution of plots utilized for the overstory and midstory plots (24 pairs of plots in 12 stands) compared to the sapling and seedling plots (12 paired plots in 4 stands).

Overstory and midstory vegetation in the riparian areas were relatively similar to that in the nonriparian areas. Ellenberg's and Sorensen's indices indicated that the

Table 3—Riparian and nonriparian overstory importance values

Species	Riparian	Nonriparian
Carya texana Buckl.	1.6	0.7
C. tomentosa (Poir.) Nutt.	3.0	0.7
Fraxinus pennsylvanica (Borkh.) Sarg.	1.1	
Liquidambar styraciflua L.	5.6	0.7
Nyssa sylvatica Marsh.	1.7	1.3
Pinus echinata Mill.	59.3	78.7
P. taeda L.	1.1	
Quercus alba L.	9.3	7.1
Q. falcata Michx. var *falcata*	2.3	0.8
Q. marilandica Muenchh.		0.6
Q. rubra L.	1.5	0.6
Q. stellata Wangenh.	8.9	6.2
Q. velutina Lam.	1.8	2.6
Ulmus alata Michx.	1.8	
U. americana L.	0.6	
U. rubra Muhl.	0.6	

Table 4—Riparian and nonriparian midstory importance values

Species	Riparian	Nonriparian
Acer rubrum L.	4.7	0.3
Amelanchier arborea (Michx. f.) Fern.	0.5	0.1
Carya texana Buckl.	8.4	3.6
C. tomentosa (Poir.) Nutt.	7.4	3.6
Cornus florida L.	1.5	4.3
Hardwood misc.	1.3	
Juniperus virginiana L.	0.9	1.3
Liquidambar styraciflua L.	5.2	2.4
Nyssa sylvatica Marsh.	5.2	3.0
Ostrya virginiana (Mill.) K.Koch	1.1	
Pinus echinata Mill.	18.0	28.6
Prunus serotina Ehrh.	0.5	0.1
Quercus alba L.	11.7	16.0
Q. falcata Michx. var *falcata*	2.1	3.2
Q. marilandica Muenchh.	0.3	6.2
Q. stellata Wangenh.	16.7	9.2
Q. rubra L.	1.8	0.4
Q. velutina Lam.	2.7	10.0
Ulmus alata Michx.	9.1	7.5
U. americana L.	0.3	
U. rubra Muhl.	0.7	0.2

Table 5—Riparian and nonriparian sapling importance values

Species	Riparian	Nonriparian
Acer rubrum L.	9.2	10.2
Carya texana Buckl.	2.6	7.0
C. tomentosa (Poir.) Nutt.	1.3	4.2
Chionanthus virginicus L.	1.4	
Cornus florida L.	11.4	12.9
Crataegus marshallii Eggl.	1.5	
Diospyros virginiana L.	1.0	1.2
Fraxinus pennsylvanica (Borkh.) Sarg.	1.6	
Juniperus virginiana L.	1.8	1.6
Liquidambar styraciflua L.	10.8	3.4
Ostrya virginiana (Mill.) K.Koch	19.5	11.5
Nyssa sylvatica Marsh.	12.8	11.2
Pinus echinata Mill.	2.2	4.3
P. taeda L.	1.6	
Prunus serotina Ehrh.	1.6	1.2
Quercus alba L.	4.1	9.1
Q. marilandica Muenchh.	1.0	2.5
Q. stellata Wangenh.	1.2	7.5
Rhamnus caroliniana Walt.		1.8
Ulmus alata Michx.	12.1	5.9
Vaccinium arboreum Marsh.	1.0	4.3
Vitis rotundifolia Michx.	0.8	

Table 6—Riparian and nonriparian sapling importance values

Species	Riparian	Nonriparian
Acer rubrum L.	3.5	15.0
Carpinus caroliniana		0.9
Carya texana Buckl.	0.9	5.6
C. tomentosa (Poir.) Nutt.	1.1	
Cercis canadensis L.		0.9
Cornus florida L.	14.9	26.1
Crataegus marshallii Eggl.	0.6	
Fraxinus pennsylvanica (Borkh.) Sarg.	3.7	
Hardwood misc.	2.3	
Juniperus virginiana L.	0.9	0.9
Liquidambar styraciflua L.	1.0	
Nyssa sylvatica Marsh.	14.9	7.6
Ostrya virginiana (Mill.) K. Koch	24.9	4.1
Prunus serotina Ehrh.	1.7	
Quercus alba L.	13.1	8.1
Q. marilandica Muenchh.	0.6	4.3
Q. phellos L.	1.0	0.9
Q. rubra L.	0.9	1.5
Q. stellata Wangenh.	5.7	8.1
Q. velutina Lam.	6.3	3.4
Rhamnus caroliniana Walt.	7.5	5.5
Rhus aromatica Ait	1.7	
Ulmus alata Michx.	0.3	3.2

overstory was respectively 84.8 percent and 66.6 percent similar while the midstory was 74.4 percent and 71.8 percent similar. The high degree of similarity in these two areas could be attributed to the incised nature of the ephemermal stream corridors. The borders of the SMZ in these upland pine-hardwood stands contain short but steep slopes. These slopes are relative dry and most likely similar to those in the upland areas. Distribution of plant communities on either side of streams usually reflects the gradient of environmental factors such as water availability which dominates these areas (Hancock and others 1996). However, these slopes increase drainage as well as elevate trees above standing water during storm events. These factors create a drier environment more suitable to upland species such as shortleaf pine, black hickory (*Carya texana* Buckl.), mockernut hickory [*C. tomentosa* (Poir.) Nutt.], post oak, white oak, black oak (*Q. velutina* Lam.), and blackjack oak (*Q. marilandica* Muenchh) than species typically found in riparian landscapes. This suggests that more hydric conditions exist only in close proximity to the stream channel and for only short periods of time. Thus stream effects on at least these two size classes of vegetation are spatially and temporally limited in these stream corridors.

Similarity indices for saplings or seedlings size classes were much lower than those calculated for the overstory or midstory size classes. Similarities of 27.1 percent and 18.5 percent were calculated using Sorensen's index for the saplings and seedlings respectively. Ellenberg's index indicated a 32.5 percent similarity between riparian and nonriparian saplings. Reduction in vegetation similarity of the seedling and sapling size classes compared to that of the midstory and overstory may again reflect the differences in the number of plots utilized for index calculation of each group.

However, several species in the seedling and sapling size classes occur on the riparian plots that do not occur in the nonriparian plots and one or two species occur in the nonriparian plots that do not occur in the riparian plots. Many of the saplings and seedlings occuring in these areas are considered late successional species within these ecosystems. Late successional species are typically resource specialists (Odum 1997). Compared to early successional species in the larger size class, these late successional species may better reflect subtle differences in environmental conditions within or outside the influence of these streams, thus creating the observed differences in similarity among size classes.

Composition

Shortleaf pines dominated the overstory with importance values of 59.3 percent and 78.7 percent for the riparian and nonriparian areas respectively (table 3). These two areas are primarily composed of species that are shade intolerant, favor relative dry habitats, and are intolerant of inudation (Hook 1984). No one species dominated either the riparian or nonriparian midstory. The importance value of any one species never exceeded 29 percent (table 4). Shortleaf pines and oaks (post, white and black) were major components of both riparian and nonriparian areas. Black oak and blackjack oak are two species that were prevalent in the nonriparian midstory. The midstories of both areas were comprised of intolerant to intermediate shade tolerant species but some shade tolerant species such as flowering dogwood (*Cornus florida* L.) and ironwood (*Ostrya virginiana* Mill.) also occurred. The sapling and seedling size classes were equally dominated by six to seven species (table 5). White oak was the only species that had importance values greater than 10 percent within the seedling or sapling size classes as well as in the midstory and overstory size classes. Species such as flowering dogwood, ironwood, blackgum, and red maple (*Acer rubrum* L.) which were only minor components in the overstory and midstory were major components of the woody vegetation communities in these smaller size classes (table 5). Species which are shade intolerant generally had low importance values while more shade tolerant species had high importance values in the sapling and seedling size classes (tables 5 and 6). These changes in composition reflected the lower light levels within these forest strata.

CONCLUSION

Woody vegetation within SMZ bordering ephemeral streams within shortleaf pine-hardwood stands generally was found to be significantly more diverse and have a greater species richness than woody vegetation growing outside the influence of these streams. Differences in diversity and richness between riparian and nonriparian areas were generally greater in the larger than smaller size classes. The density of the overstory woody vegetation was greater outside than inside these SMZ but did not significantly differ in the smaller size classes. The riparian vegetation communities in these SMZ appear to contribute to the overall diversity of the landscape. However, due to the relatively small size of these streams and their relatively minor impact on moisture regimes, vegetation communities are much more similar to upland communities than would be expected for riparian communities growing along perrenial streams with larger floodplains.

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Session 2
Phase II Wildlife Research

Larry D. Hedrick, Moderator

INITIAL SMALL MAMMAL RESPONSES TO ALTERNATIVE PINE REGENERATION METHODS IN ARKANSAS AND OKLAHOMA: PRELIMINARY FINDINGS

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Abstract—We studied winter small mammal communities in the Ouachita Mountains of west-central Arkansas and eastern Oklahoma using Sherman live traps in four replications of four regeneration treatments (clearcut/plant, shelterwood, single-tree selection, and group selection), plus four mature, untreated "controls." Data on relative small mammal abundance, species richness, and diversity after one, three, and five growing seasons after harvest are presented. Capture success (all species included) generally peaked in all harvested treatments after the first growing season, declined markedly thereafter, but remained 2.6 (single tree) to 4.1 (clearcut) times greater than in controls five growing seasons after harvest. Total small mammal abundance in clearcut and shelterwood stands was significantly higher ($P \leq 0.05$) than in controls all years. With data for all species included, species richness and diversity did not differ among harvested stands in any year, but shelterwood stands had higher richness and diversity than controls in the third growing season.

INTRODUCTION

Small mammals play key roles in ecosystem processes. They occupy important trophic levels of the food web, often serving both as predators and as prey for many vertebrates. Fungus-consuming species are largely responsible for dispersal of hypogeous fungal spores that form mycorrhizae, which are required by most higher plants for optimum growth and health (Maser and others 1978). Many species of small mammals consume detrimental insects (for example, see Elkinton and others 1996) and aid in seed dispersal. Their digging and burrowing improve soil aeration and friability, and can significantly influence forest hydrology (Ursic and Esher 1988).

Consumption of pine seeds by small mammals can adversely affect pine regeneration (Pank 1974, Smith and Aldous 1947, Stephenson and others 1963), but impacts likely vary from trivial during years of bumper seed crops to devastating in years of below average seed production (Wittwer and Shelton 1992). For shortleaf pine (*Pinus echinata* Mill.) in the Ouachita and southern Ozark Mountains, seed production typically follows a feast or famine pattern with roughly one-third of the seed crops "good" or "better" (≥80,000 sound seeds per acre) (Shelton and Wittwer 1996). Seed production varies widely from year to year, and long periods of low seed production may occur, especially in the drier western portions of the Ouachita Mountains. Given this variability in shortleaf pine seed production, small mammal population responses to silvicultural practices that rely on natural regeneration are of paramount importance.

In response to growing public concern over clearcutting and hardwood control practices on national forests in Arkansas and Oklahoma, a long-term, multidisciplinary, stand-level, research and demonstration project was begun in the Ouachita Mountains in 1991 (Baker 1994). A primary objective of this research is to compare effectiveness of different partial cutting methods for natural regeneration of shortleaf pine relative to clearcutting and planting. As part of this multidisciplinary project, we are evaluating temporal changes in small mammal and bird communities and habitat conditions under these cutting treatments, which were implemented during the summer of 1993. We studied small mammal populations during two winters prior to harvesting (Tappe and others 1994). Here we summarize preliminary findings on small mammal responses to alternative pine regeneration methods at the ends of the first, third, and fifth growing seasons after harvest.

METHODS

Study Areas

Four replications of five treatments, blocked by physiographic zones (Baker 1994), were randomly assigned to 20 stands located on 9 districts of the Ouachita National Forest and the southernmost district of the Ozark-St. Francis National Forest (Thill and others 1994). All stands were late-rotation (≥60 years old), ≥35 acres in size (and blocky in shape to minimize confounding edge influences), and occupied predominantly south, southeastern, or southwestern aspects.

All stands contained ephemeral, or occasionally intermittent, stream drainages that typically flow only for short periods after heavy storms. Unharvested buffer strips ("greenbelts" henceforth) were retained along these drainages for watershed protection. Ephemeral and intermittent greenbelts were typically 65 and 130 feet wide, respectively. Greenbelts comprised 4 to 20 percent of stand acreage and averaged 10.9 percent across all 16 harvested stands.

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Pretreatment conifer basal area for all 20 stands averaged 76.7 square feet per acre and consisted almost entirely of shortleaf pine and a few eastern redcedars (*Juniperus virginiana* L.); hardwood basal area averaged 36.6 square feet per acre (Thill and others 1994). Stand conditions immediately after treatment are described by Thill and others (2000).

For additional information on study areas, climate, geology, stand selection, and experimental design, see Baker (1994) and Thill and others (2000). For information on pretreatment stand conditions and wildlife habitat characteristics, see Guldin and others (1994) and Thill and others (1994), respectively.

Treatments

Although the larger study involves 13 treatments, we chose a subset of treatments where an overstory hardwood component was retained to improve wildlife habitat and aesthetics. We also included clearcutting, which had been the principal Forest Service pine regeneration method in the Ouachitas for decades. Four treatments (clearcut, shelterwood, single-tree selection, and group selection) plus untreated controls were compared. Harvesting was completed between June 1 and September 30, 1993; site preparation, where needed, was conducted during the winter of 1993-94.

Clearcut—All merchantable pines and hardwoods were harvested except for 2 to 5 square feet per acre of hardwood basal area retained for mast production and/or cavity and den sites. All trees not harvested or retained were injected with herbicide (Baker 1994). After harvesting, all clearcuts were to be mechanically ripped to facilitate pine planting, but contractors could not be located to rip two stands. With this exception, all clearcuts were treated identically (Thill and others 2000). Genetically improved shortleaf pines were planted on an 8- by 10-foot spacing between December 1994 and March 1995. One stand was replanted in February 1996 due to inadequate stocking.

Shelterwood—From 20 to 40 of the largest pines and hardwoods per acre were retained. The basal area of trees that were left was 30 to 40 square feet per acre, of which 5 to 15 square feet per acre were overstory hardwoods. All other pines and hardwoods were harvested or felled and left on the ground.

Single-tree selection—Partial harvest of pines and hardwoods resulted in residual basal areas of 45 to 65 square feet per acre. From 5 to 20 square feet per acre of the residual basal area was in hardwoods. Site preparation consisted of removing all hardwoods <5.9 inches diameter at breast height (d.b.h.).

Group selection—Approximately 10 percent of each stand was clearcut in openings generally ranging from 0.5 to 2.0 acres in size. Most pines were cut within these openings, but 5 to 10 square feet per acre of overstory hardwood basal area was retained. Within the matrix surrounding these openings, pines were thinned to 70 to 80 square feet per acre basal area, but no hardwoods were removed. Site preparation in group openings consisted of chain-saw felling all hardwoods <5.9 inches d.b.h.; no site preparation occurred in the surrounding matrix.

Untreated controls—These areas supported second-growth, late rotation (62 to 76 years old), largely even-aged, pine-hardwood stands. Management consisted of protection from wildfire and insects.

Transects/Trap Stations

Eighty permanent trap stations were established along transects oriented parallel to topographic contours of each stand. No two transects were closer than 98 feet, trap stations were at least 49 feet apart, and no station was closer than 164 feet from stand boundaries (Thill and others 1994).

Trapping

Small mammals were trapped for seven consecutive nights between January 5-14 in 1995, 1997, and 1999. Consequently, our data characterize small mammal communities present one, three, and five full growing seasons after harvesting. Two Sherman live traps (3.0 by 3.5 by 9.0 inches) were placed at each of the 80 trap stations in each stand to ensure opportunities for multiple captures per trap station. These traps are sufficient to capture mammals as small as southern short-tailed shrews (*Blarina carolinensis*) and as large as eastern woodrats (*Neotoma floridana*); however, we were unable to adjust trigger sensitivity enough to consistently capture least shrews (*Cryptotis parva*). Traps were placed on bare ground within 16.4 feet of each station center, and (where possible) adjacent to down logs, burrows, stumps, and rocks to increase trap success. At least eight (10 percent) of the trap stations were placed within greenbelts if transects crossed sufficient greenbelt area.

Traps were baited with rolled oats, and cotton was placed in each trap to minimize trap mortality. Captured mammals were marked and released at the site of capture after recording species, sex (when possible), and trap location.

Small mammal nomenclature follows Sealander and Heidt (1990). Because accurate separation of our sympatric *Peromyscus* species is not possible in the field (Laerm and Boone 1994, Rich and others 1996), all white-footed mice (*Peromyscus leucopus*), cotton mice (*P. gossypinus*), Texas mice (*P. attwateri*), and deer mice (*P. maniculatus*) were grouped as *Peromyscus* spp.

Analyses

We computed captures per 100 trap nights as an index of relative abundance. Total available trap nights were computed by adjusting for empty sprung traps; traps that contained recaptured animals were also considered unavailable. An index of species richness was calculated as the number of species encountered on each area over the 7-day trapping period. Species diversity (Shannon-Weiner diversity index) was based on these composited data (Magurran 1988). Differences among treatments for all variables were evaluated using one-way ANOVAs (n = 20 stands). We used REGWQ multiple range tests (alpha = 0.05) to separate means (SAS Institute Inc. 1988). Analyses were run with and without *Peromyscus* data to ensure that the relatively high abundance of this group was not masking treatment effects. Variances of new captures per 100 trap nights were heterogeneous in 1995 with *Peromyscus* data included and in 1997 and 1999 with *Peromyscus* excluded; in each of these cases, values were rank transformed prior to analyses of variance. Except where noted, presented results include *Peromyscus* data.

Posttreatment surveys sampled a mixture of habitat conditions within the 16 treated stands included treated openings and thinned areas, temporary logging roads, and greenbelts. These differences within treated areas are ignored and small mammal responses to operational, stand-level conditions are reported here.

RESULTS AND DISCUSSION

Trap Success

Small mammal captures (excluding recaptures) across all treatments totaled 1,501; 1,151; and 1,091 during 1995, 1997, and 1999. Total trap nights, adjusted for recaptures and empty sprung traps, were 20,560; 20,774; and 19,131; respectively, for 1995, 1997, and 1999. Trap success across all 20 stands declined from an average of 7.3 captures per 100 trap nights in 1995 to 5.5 in 1997 and 5.7 in 1999.

Relative Abundance

Approximately 1.5 years after harvest (January 1995), small mammal trap success was about 3.0 to 5.5 times higher in harvested stands than in controls (fig. 1). Clearcut and shelterwood stands had higher capture rates than controls

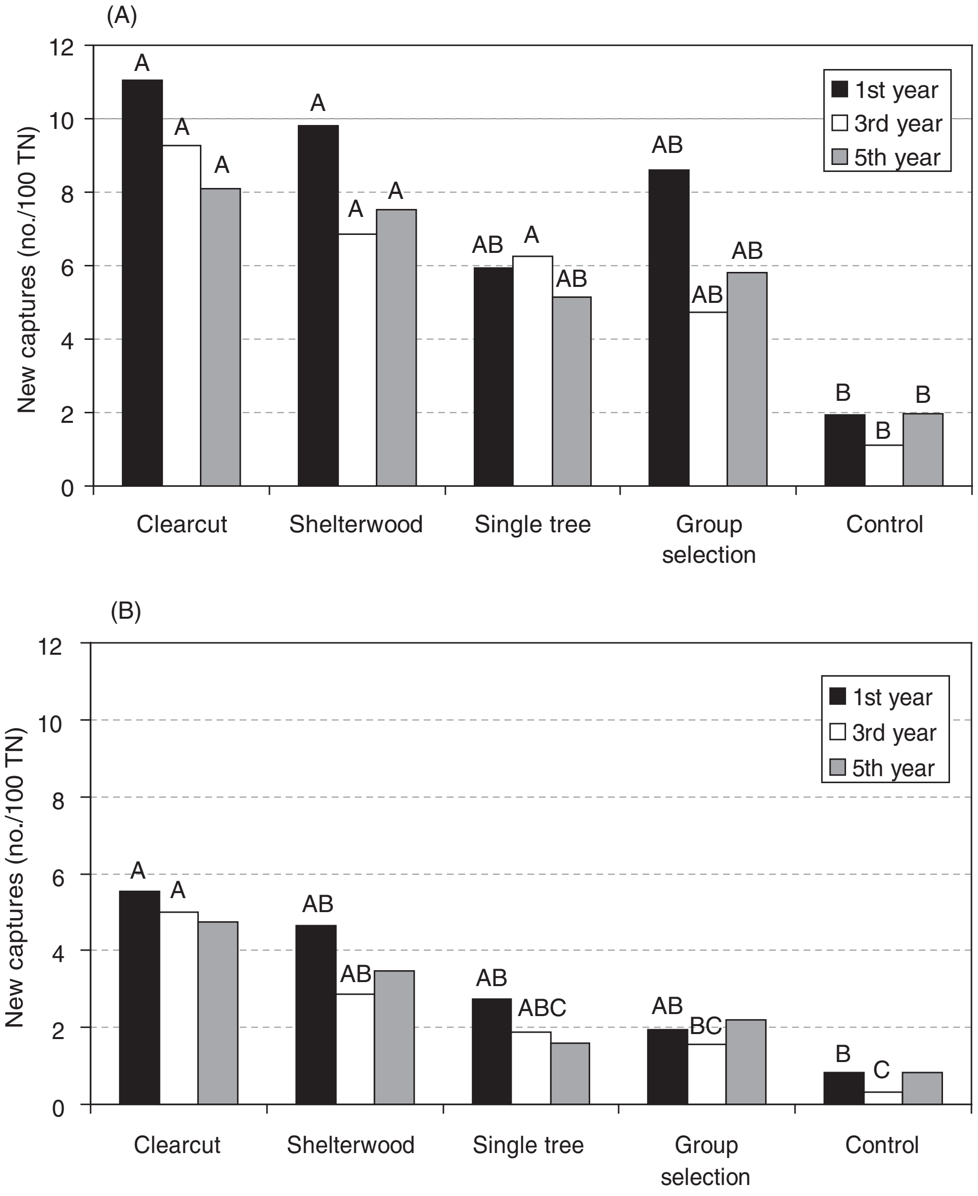

Figure 1—Relative abundance of small mammals (mean number of new captures per 100 trap nights, all species combined) by treatment following the first, third, and fifth growing seasons after harvesting pine-hardwood forests in the Ouachita Mountains of Arkansas and Oklahoma. Within years, bars with different letters are significantly different ($P \leq 0.05$). Top graph (A) includes *Peromyscus* spp.; bottom graph excludes *Peromyscus* spp.

all three sampling years; however, except for single-tree selection in 1997, differences between single-tree and group selection stands and controls were not significant (*P*>0.05) (fig. 1). Among the four harvested treatments, small mammal abundance was highest the first year after harvest, except in single-tree selection stands. When data for the four harvested treatments were averaged within years, small mammal abundance was 4.6, 6.2, and 3.5 times greater than in controls during 1995, 1997, and 1999, respectively.

With *Peromyscus* data excluded, clearcuts had significantly greater relative abundance of small mammals than the controls during 1995. In 1997, clearcut and shelterwood stands had significantly greater small mammal abundance than controls; clearcuts also had greater (*P*<0.05) abundance than group selection stands (fig. 1). By 1999, none of the treatment differences were significant.

Species Richness

Ten taxa were captured, but five (*Peromyscus* spp., *Reithrodontomys fulvescens, Blarina carolinensis, Ochrotomys nuttalli,* and *Neotoma floridana*) consistently comprised over 95 percent of captures during all three sampling years after treatment (table 1). Except for two least shrews captured in one clearcut during 1997, all species trapped after harvest were also encountered prior to treatment (Tappe and others 1994).

Small mammal species richness did not differ (*P*>0.05) among treatments in 1995 or 1999 (fig. 2). The only significant difference occurred in 1997, when richness was 2.4 times greater in shelterwood stands than in controls. *Peromyscus* spp. were present on all areas; excluding Peromyscus did not change the relationships among the treatments.

Diversity

As with species richness, there were generally no differences in small mammal diversity among treatments (fig. 3). The only significant difference occurred in 1997 when diversity was 2.2 times greater in shelterwood than in control stands. With *Peromyscus* data excluded, single-tree stands were more diverse than control stands in 1997.

CONCLUSIONS

Prior to treatment, study areas were characterized by high canopy coverage, an abundance of relatively small diameter midstory and overstory hardwoods, and limited understory browse and herbage (Thill and others 1994). These untreated, late-rotation stands also were characterized by low small mammal abundance, species richness, and diversity (Tappe and other 1994). We suspect this condition is related, at least partially, to limited soft and hard mast availability [Perry and others, in press (a); in press (b)]. Following harvest, these forage items increased rapidly.

Table 1—Nomenclature, trapping effort, total winter captures, and percent composition of small mammals by species/species group following the first (1995), third (1997), and fifth (1999) growing seasons after harvesting of pine-hardwood stands in the Ouachita Mountains of Arkansas and Oklahoma[a]

		1995		1997		1999	
Nomenclature	Common name	Captures	Percent	Captures	Percent	Captures	Percent
Order Insectivora							
Family Soricidae							
Blarina carolinensis	Southern short-tailed shrew	114	7.6	95	8.2	115	10.5
Cryptotis parva	Least shrew	0	0.0	2	0.2	0	0.0
Order Rodentia							
Family Muridae							
Microtus pinetorum	Woodland vole	5	0.3	31	2.7	33	3.0
Mus musculus	House mouse	0	0.0	0	0.0	0	0.0
Neotoma floridana	Eastern woodrat	82	5.5	71	6.2	101	9.3
Ochrotomys nuttalli	Golden mouse	132	8.8	53	4.6	70	6.4
Oryzomys palustris	Marsh rice rat	5	0.3	0	0.0	0	0.0
Peromyscus spp.[b]	Mice	866	57.7	676	58.7	609	55.8
Reithrodontomys fulvescens	Fulvous harvest mouse	277	18.4	213	18.5	143	13.1
Sigmodon hispidus	Hispid cotton rat	19	1.3	9	0.8	5	0.5
Family Sciuridae							
Glaucomys volans	Southern flying squirrel	1	0.1	0	0.0	15	1.4
Tamias striatus	Eastern chipmunk	0	0.0	1	0.1	0	0.0
Total captures:		1,501		1,151		1,091	
Total trap nights (TN)[c]:		20,560		20,774		19,131	
Total captures/100TN:		7.3		5.5		5.7	

TN = trap nights.

[a] Logging within the 16 harvested stands occurred during summer 1993. Data from all 20 stands are combined.

[b] Includes *P. leucopus*, *P. gossypinus*, *P. maniculatus*, and *P. attwateri*.

[c] Trap nights were adjusted for recaptures and sprung/empty traps; total unadjusted trap nights was 22,400 per year or 1,120 stand per year.

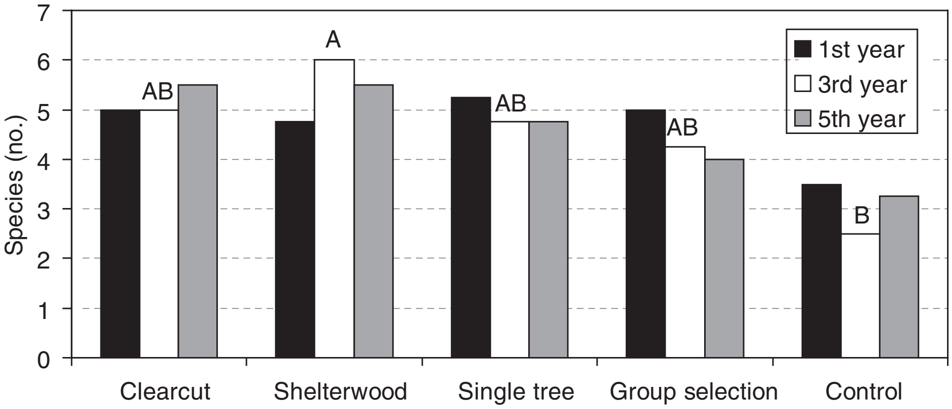

Figure 2—Small mammal species richness by treatment following the first, third, and fifth growing seasons after harvesting pine-hardwood forests in the Ouachita Mountains of Arkansas and Oklahoma. Within years, bars with different letters are significantly different ($P \leq 0.05$). Statistical results are similar with and without *Peromyscus* spp.

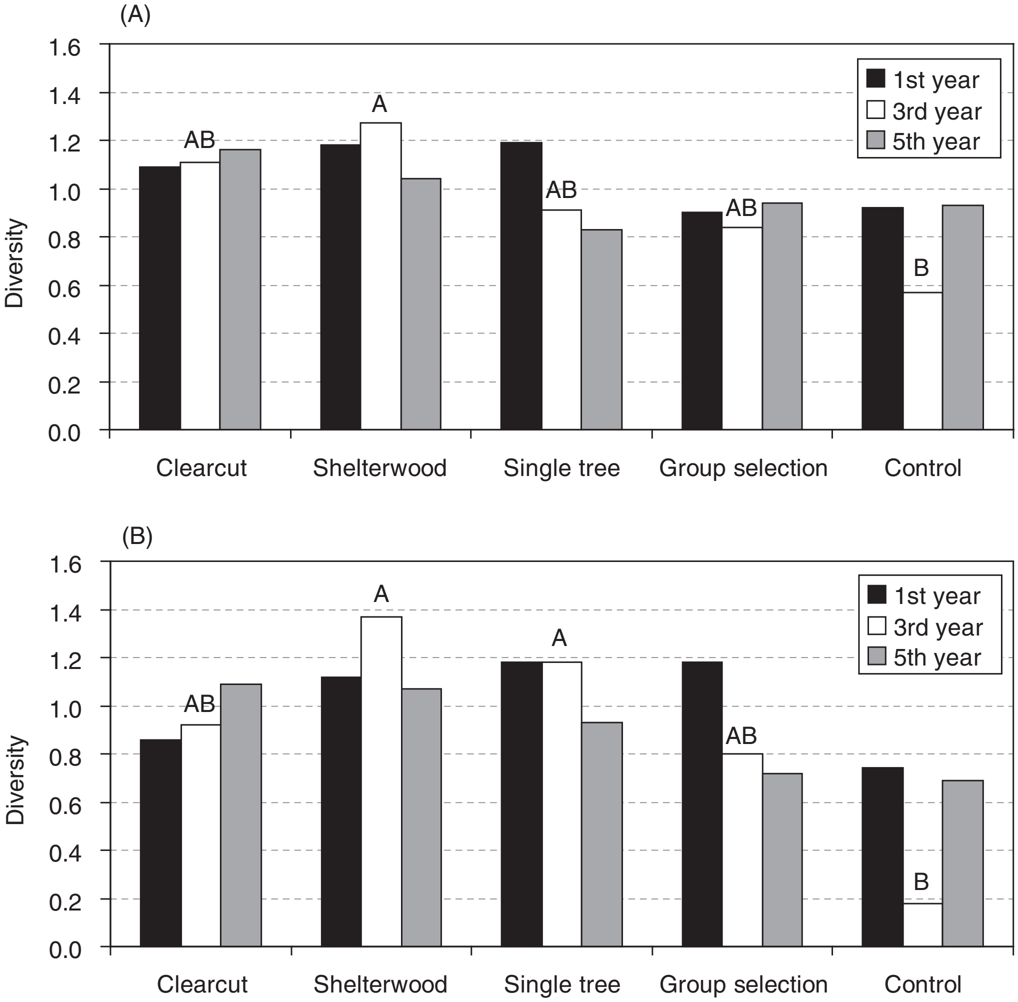

Figure 3—Small mammal species diversity by treatment following the first, third, and fifth growing seasons after harvesting pine-hardwood forests in the Ouachita Mountains of Arkansas and Oklahoma. Within years, bars with different letters are significantly different ($P \leq 0.05$). Top graph (A) includes *Peromyscus* spp.; bottom graph excludes *Peromyscus* spp.

Single-tree and group-selection stands, largely even-aged at study initiation, are in transition to an uneven-aged stand structure. Before these stands attain this structure, they must be harvested additional times and obtain satisfactory pine regeneration and survival. Consequently, it is too soon to draw definitive conclusions about small mammal responses to even- and uneven-aged regeneration methods. However, during the early stages of this transition, our data suggest that both uneven-aged treatments should yield similar small mammal species richness and diversity to those yielded by the two even-aged treatments on sites similar to those studied here. Likewise, although small mammal abundance was somewhat lower in single-tree and group- selection stands than in clearcut or shelterwood stands in all 3 years, these differences were not statistically significant. Additionally, single-tree and group-selection stands both had substantially (though not always statistically) higher small mammal abundance than unharvested stands.

From a pine regeneration standpoint, small mammal numbers increased rapidly and were at their highest level soon after harvesting. Because of logging and site preparation disturbances, seedbed conditions are also typically optimal for natural pine regeneration soon after harvesting. Given these conditions, the seed production characteristics of shortleaf pine, and the rapid response of small mammals to harvesting, foresters have legitimate concerns about seed consumption by small mammals (and other wildlife). Thus, additional research investigating actual impacts under these silvicultural systems may be advisable. Increases in bird abundance following these treatments were not as rapid as for small mammals (Thill and others 2000).

Small mammal abundance, richness, and diversity did not differ (P>0.05) between clearcut and shelterwood stands during any sampling year with or without inclusion of *Peromyscus* data. Thus, shelterwood regeneration methods can achieve similar small mammal responses as clearcutting within aesthetically more-pleasing settings (Personal communication. Victor A. Rudis. 2000. Research Forester, USDA Forest Service, Southern Research Station, Forestry Sciences Laboratory, 201 Lincoln Green, Starkville, MS 39759) during at least the first 5 years after harvest. Healthy small mammal populations should benefit a host of vertebrate predators.

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INITIAL BIRD RESPONSES TO ALTERNATIVE PINE REGENERATION METHODS IN ARKANSAS AND OKLAHOMA: PRELIMINARY FINDINGS

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Philip A. Tappe, and David G. Peitz[1]

Abstract—We studied spring songbird communities in a predominantly forested landscape in the Ouachita Mountains of west-central Arkansas and eastern Oklahoma. Relative bird abundance, species richness, diversity, and evenness values are presented for the first, third, and fifth years following harvesting in four replications of four regeneration treatments (clearcut/plant, shelterwood, single-tree selection, and group selection), plus mature, untreated “controls.” Within clearcut and shelterwood stands, relative abundance, species richness, and diversity of birds increased dramatically from the first to the third year, with smaller increases from the third to the fifth year. Single-tree and group selection stands followed this same pattern for species richness and diversity, but not for relative bird abundance. Within years, differences in response to harvesting generally were not significant, but group selection stands typically had the lowest values for the four harvesting treatments. Of the five treatments, controls typically had the lowest response variable values.

INTRODUCTION

Even-aged silviculture employing clearcutting, site preparation, and planting of pines has dominated forest management practices on forest industry and national forest lands throughout the South for the last 30 years (Baker 1989). Although young pine plantations provide excellent habitat for wildlife species adapted to early seral stages, even-aged pine silviculture, especially under short rotations, can diminish critical habitat for wildlife that require an abundance of snags and cavity trees, hardwoods, hard mast, and large down woody material.

Under recently adopted principles of ecosystem management (Guldin 1996, Overbay 1992), the USDA Forest Service is evaluating the effects of even- and uneven-aged regeneration methods on both an experimental and operational basis (for example, see Baker 1994, Kitchens 1989). One such project was initiated in Arkansas in 1991 on the Ouachita National Forest, the largest national forest (nearly 1.8 million acres) in the southern region. This multiphase project is evaluating partial cutting methods, some of which include long-term retention of a mixed-species overstory, employing natural regeneration as alternatives to clearcutting and planting (Baker 1994).

Phase II of this initiative is a stand-level, replicated study comparing an array of forest outputs and outcomes under 12 silvicultural treatments plus untreated controls (Baker 1994). Two years of pretreatment bird and habitat data were collected in 20 of these stands in 1992 and 1993 (Petit and others 1994). Stands were harvested during summer 1993, site preparation was conducted during the 1993-94 dormant season, and posttreatment bird surveys were initiated during May 1994. This paper reports breeding bird populations 1, 3, and 5 years after treatment and compares them with populations in untreated control stands.

METHODS

Study Areas

The 52 stands that comprise the larger study were randomly selected from a list that included all mixed shortleaf pine (*Pinus echinata* Mill.)-hardwood stands from randomly selected township and range strips in the Ouachita and Ozark National Forests that were candidates for regeneration. The stands also met the following pretreatment criteria: ≥ 60 years old; ≥ 35 acres in size (and blocky in shape to minimize confounding edge influences); predominantly southerly (south, southwest, or southeast) aspect; 60 to 110 square feet per acre pine basal area; and 20 to 50 square feet per acre hardwood basal area (Baker 1994). These 52 stands were blocked into 4 physiographic zones with 13 stands per block. Within each block, treatments were randomly assigned to the stands, yielding a completely randomized block design (Baker 1994). Because of limited resources, we chose to study only 5 of the 13 treatments. Thus, data reported here are from 20 (4 replications of 5 treatments) of the 52 stands (Thill and others 1994).

Eighteen of our study areas were on the Ouachita National Forest and two were on the southernmost district of the Ozark-St. Francis National Forest. Study areas were 35 to 40 acres in size and had slopes ranging from 0 to 15 percent. They generally occupied relatively xeric sites (southerly aspects, usually of mid- or lower slope position) characterized by high canopy coverage, an abundance of mostly small midstory and overstory hardwoods, and limited understory browse and herbage prior to treatment (Thill and others 1994). Pretreatment conifer basal area averaged 76.7 square feet per acre and consisted almost entirely of shortleaf pine and a few eastern redcedars (*Juniperus virginiana* L.); hardwood basal area averaged 36.6 square feet per acre (Thill and others 1994). Common hardwoods included oaks (*Quercus* spp.), hickories (*Carya* spp.), winged elm (*Ulmus*

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alata Michx.), and black gum (*Nyssa sylvatica* Marsh.) (Guldin and others 1994). Pine site index (base age 50) averaged 62.2 feet over all 52 stands. For information on topography, soils, and climate, see Baker (1994).

All 20 stands contained ephemeral, and occasionally intermittent, drainages that typically flow only during and after heavy rains. Unharvested linear strips ("greenbelts") were established along these drainages in the 16 treated stands for watershed protection. Ephemeral and intermittent greenbelts were typically 65- and 130-feet wide, respectively. Greenbelts comprised 4 to 20 percent of the 16 harvested stands and averaged 10.9 percent. Prior to treatment, there was generally little difference in woody vegetation composition or structure between greenbelts and the surrounding habitat. On the upper slopes of most stands, these drainages tend to be steep and rocky and typically contain water only for brief periods.

Postharvest overstory basal area data were obtained as described by Guldin and others (1994). For additional information on sites, treatments, general stand characteristics, and pretreatment wildlife habitat conditions, see Baker (1994), Guldin and others (1994), and Thill and others (1994), respectively.

Treatments

Although the larger study involves 13 treatments, we chose to evaluate those treatments where an overstory hardwood component was retained to improve wildlife habitat and aesthetics; we also included clearcutting, which had been the principle timber harvesting and regeneration method for decades. Thus, four treatments (clearcut, shelterwood, single-tree, and group selection) plus an untreated control were evaluated. Harvesting was completed between June 1 and September 30, 1993.

Clearcut—All merchantable pines and hardwoods were harvested except for 2 to 5 square feet per acre of hardwood basal area retained for mast production and/or cavity and den sites. All trees that were not harvested or retained were injected with herbicide (Baker 1994). After harvesting, all four sites were scheduled to be ripped on the contour at 10-foot intervals to a depth of 6 to 8 inches and planted with genetically improved shortleaf pine seedlings on an 8-foot spacing within the rips. However, contractors could not be obtained to rip two sites; these sites were treated identically to the other two clearcuts but were not ripped. Genetically improved pines were planted between December 1994 and March 1995. One stand was replanted in February 1996 due to inadequate stocking.

Shelterwood—From 20 to 40 of the largest pines and hardwoods per acre were retained; the combined basal area of trees that were left was 30 to 40 square feet per acre, of which 5 to 15 square feet per acre were hardwoods. All other pines and hardwoods were harvested or felled and left on the ground.

Single-tree selection—Under this treatment, some pines and hardwoods will be harvested approximately every 10 years. As with future thinning treatments, residual basal area after this first cutting cycle ranged from 45 to 65 square feet per acre, with 5 to 20 square feet per acre in hardwoods. Site preparation consisted of removing all hardwoods < 5.9 inches diameter at breast height (d.b.h.).

Group selection—As in the single-tree selection stands, portions of these stands will be harvested approximately every 10 years. During each harvest entry, approximately 10 percent of each stand will be cut in openings that will generally range from 0.5 to 2.0 acres in size. Because 1993 was the first entry, about 10 percent of our stands were in group openings. All pines in these openings were cut, but 5 to 10 square feet per acre of overstory hardwood basal area was retained. Within the matrix surrounding these openings, pines were thinned to 70 to 80 square feet per acre basal area, but no hardwoods were removed. Site preparation in group openings consisted of chain-saw felling all hardwoods < 5.9 inches d.b.h.; no site preparation occurred outside these openings.

Untreated controls—With the exception of protection from severe loss to wildfire or insects, no management was or will be conducted in these mature second-growth, pine-hardwood stands.

Bird Surveys

Before treatment, bird populations were surveyed in five or six (depending on stand size and shape) permanent 131-foot-radius (1.24-acres) point-count plots (Verner 1988) spaced evenly over each of the 20 study areas (Petit and others 1994). Plots were usually ≥ 425 feet apart and all plot edges were ≥ 295 feet from stand boundaries. Beginning in 1994, we randomly deleted one plot from stands where six plots had been established so that all posttreatment comparisons would be based on identical sampling effort.

Because group selection opening locations were not known when permanent sampling plots were established, posttreatment surveys sampled a mix of openings and surrounding (matrix) habitat. Likewise, posttreatment surveys also sampled some greenbelts in most harvested stands. Thus, data presented here characterize operational, stand-level conditions rather than individual, within-stand habitat components such as greenbelts, openings, or matrix habitat.

Plots were surveyed for birds once each by three observers in 1994 and once each by six observers in 1996 and 1998. All surveys were conducted between May 3 and June 12. Surveys at each plot lasted 10 minutes, and all surveys were conducted within 3.5 hours of sunrise. Surveys were not conducted during heavy rain or windy conditions. At each plot, all birds seen or heard within and outside the plot (but within the same stand) were recorded separately.

Analyses

Observers differed to some extent each year and sampling effort was increased the last 2 years; consequently, differences among years were not evaluated statistically but are reported for descriptive purposes. Because our primary interest is in differences among treatments within years, statistical tests are presented only for within-year comparisons. Findings reported here are based only on data collected within 131-foot-radius plots.

Relative bird abundance was calculated as the mean number of birds encountered on each plot across all observers. Relative species richness was calculated as the number of species encountered on each plot, composited across observers. Species diversity (Shannon-Weiner diversity index) and evenness are based on these composited values (Magurran 1988). Differences among treatments for all variables were evaluated using randomized-block analyses of variance (with physiographic zones as blocks and plots as subsamples: n = 100, four blocks X five treatments X five plots/stand). Where sampling error and experimental error were not different (alpha = 0.15), treatment effect was tested with sampling error; otherwise, experimental error was used to test for treatment differences. Mean separation was accomplished using REGWQ multiple range tests (alpha = 0.05) (SAS Institute Inc. 1988).

RESULTS AND DISCUSSION

Posttreatment Stand Characteristics

Prior to harvesting, pine, hardwood, and total basal area averaged 76.7, 36.6, and 113.4 square feet per acre, respectively. Pine and hardwood basal area were similar (*P*>0.05) among the 20 stands when grouped by future treatment (Thill and others 1994). The most intensive treatments, as indicated by residual basal area, occurred in group selection openings and clearcuts (fig. 1). Average residual hardwood basal area in shelterwood and single-tree selection stands was identical (13.5 square feet per acre), and slightly less than in group openings (17.0 square feet per acre). Residual hardwood basal area in the group selection matrix habitat was comparable (27.0 square feet per acre) to that of controls (28.8 square feet).

Relative Abundance

Within treatments, postharvest bird abundance increased progressively each year in all treatments except controls, which had slightly lower abundance in 1996 than in 1994 or 1998 (fig. 2). The amount of increase in bird abundance between years decreased in clearcuts and shelterwoods from 1996 to 1998 relative to the increase from 1994 to 1996 (fig. 2). This pattern was not apparent in the single-tree and group selection treatments. In fact, the differences in relative abundance for both single-tree and group selection stands between 1996 and 1998 were greater than between 1994 and 1996.

Differences in relative abundance among treatments were statistically significant during 1994 (*F* = 7.54; 4,80 df; *P* = 0.0001), 1996 (*F* = 5.17; 4,12 df; *P* = 0.0118), and 1998 (*F* = 27.32; 4,12 df; *P* = 0.0001). During the first year after treatment, relative abundance was significantly lower in controls than in other treatments; no differences existed among harvested stands (fig. 2). During the third year after treatment, abundance was lower in control stands than in clearcut and shelterwood stands. By the fifth after treatment, relative abundance was comparable in clearcut, shelterwood, and single-tree selection stands, but relative abundance in these three treatments was significantly higher than in group selection and control stands, both of which also differed significantly (fig. 2). When data for the four harvested treatments were averaged within years, relative abundance was 2.1, 3.1, and 2.5 times greater than in controls during 1994, 1996, and 1998, respectively.

Species Richness

In harvested stands, avian species richness increased substantially from the first to the third year for all treatments except controls; only moderate increases occurred in harvested stands between the third and fifth years after treatment (fig. 3). Species richness in control stands also increased each year, but this increase between years was more uniform than in treated stands. While the cause for apparent increases in species richness in control stands is unknown, the amount of between-year variation is only

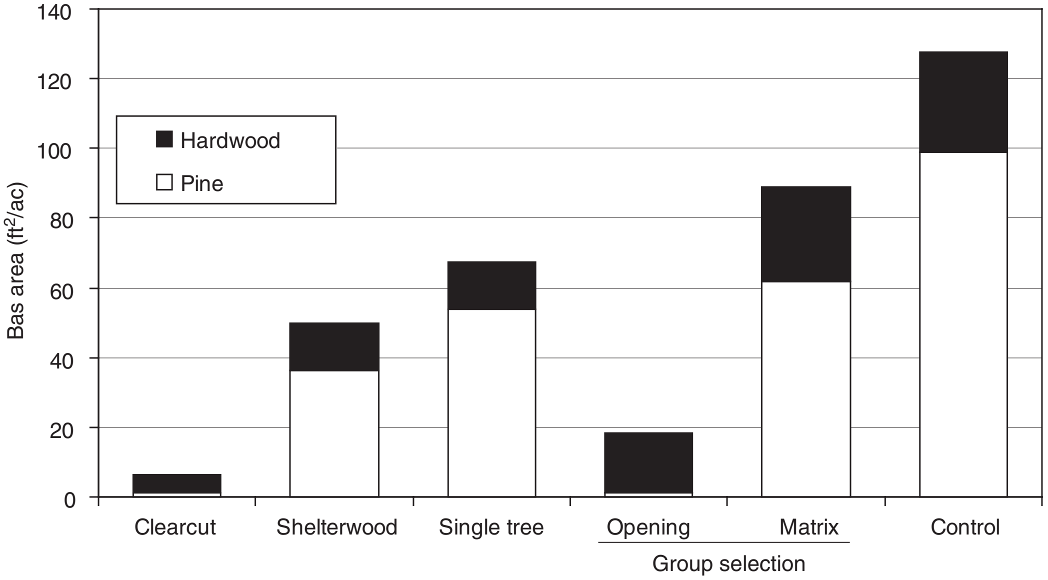

Figure 1—Pine and hardwood basal area in untreated controls and four regeneration treatments immediately after harvest of pine-hardwood stands in the Ouachita Mountains of Arkansas and Oklahoma.

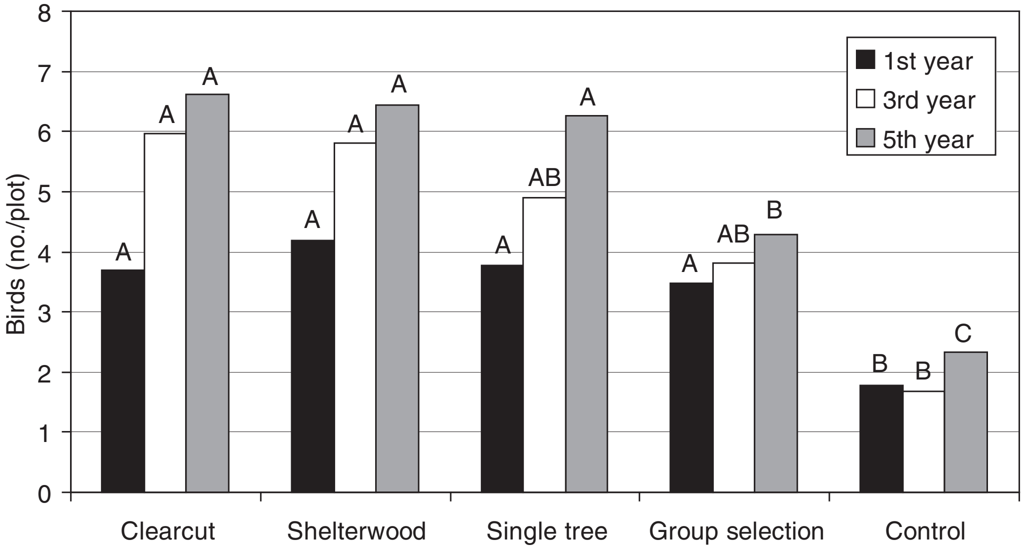

Figure 2—Relative bird abundance by silvicultural treatment during the first (1994), third (1996), and fifth years (1998) after harvesting of pine-hardwood stands in the Ouachita Mountains of Arkansas and Oklahoma. Within years, bars with different letters are significantly different ($P<0.05$).

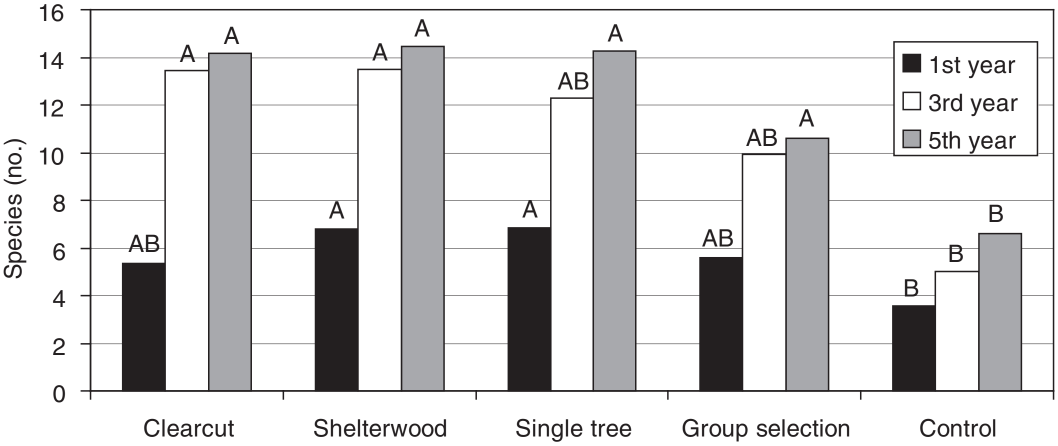

Figure 3—Bird species richness by silvicultural treatment during the first (1994), third (1996), and fifth years (1998) after harvesting of pine-hardwood stands in the Ouachita Mountains of Arkansas and Oklahoma. Within years, bars with different letters are significantly different ($P<0.05$).

slightly higher than observed prior to treatment (Petit and others 1994; fig. 6) and probably reflects inherent temporal variability and/or observer differences.

Within years, differences in species richness among treatments were significant during 1994 ($F = 3.66$; 4,12 df; $P = 0.0360$), 1996 ($F = 3.41$; 4,12 df; $P = 0.0440$), and 1998 ($F = 12.66$; 4,12 df; $P = 0.0003$). Within years, species richness was consistently lowest in untreated controls. Among the four harvested treatments, differences in richness were not significant during any year (fig. 3). However, in all 3 years species richness was significantly higher in shelterwood stands than in controls. Clearcuts and single-tree stands had higher species richness than controls in 2 years (1996 and 1998, and 1994 and 1998, respectively). Species richness in group selection stands was higher than in controls only during the fifth year. When data for the four harvested treatments were averaged within years, species richness was 1.7, 2.5, and 2.0 times greater than in controls during 1994, 1996, and 1998, respectively.

Diversity

Within harvested treatments, diversity increased each year, but the amounts of increase declined over time (fig. 4). Diversity followed a similar pattern to species richness in control stands.

Within years, controls consistently had the lowest avian diversity. Differences in diversity among the five treatments were significant during the first ($F = 4.06$; 4,12 df; $P = 0.0262$) and fifth years ($F = 8.60$; 4,12 df; $P = 0.0016$) and nearly so ($F = 3.15$; 4,12 df; $P = 0.0551$) during the third year after

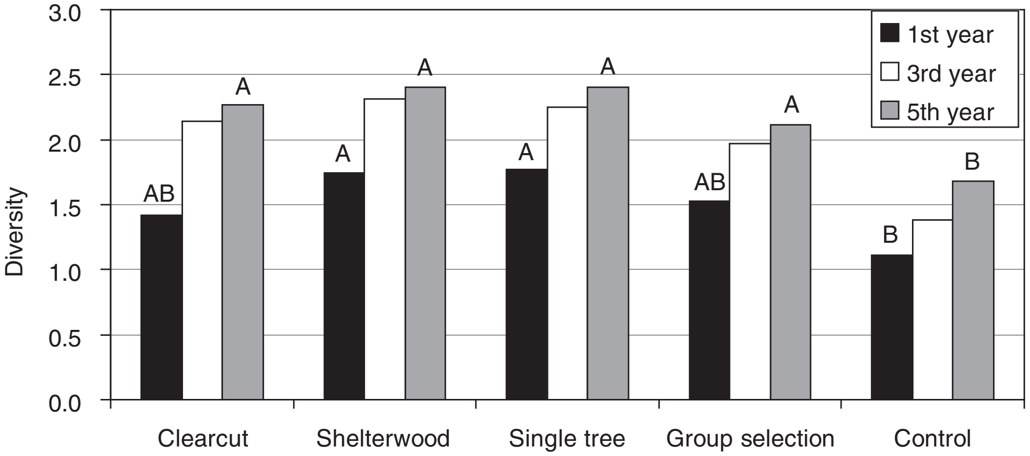

Figure 4—Bird diversity (Shannon-Weiner's index) by silvicultural treatment during the first (1994), third (1996), and fifth years (1998) after harvesting of pine-hardwood stands in the Ouachita Mountains of Arkansas and Oklahoma. Within years, bars with different letters are significantly different ($P<0.05$). No differences existed among treatments in 1996.

treatment. Bird diversity was significantly higher in shelterwood and single-tree stands than in controls the first year. By the fifth year, diversity was significantly higher in all four harvested treatments than in controls. When data for the four harvested treatments were averaged within years, avian diversity was 1.5, 1.6, and 1.4 times greater than in controls during 1994, 1996, and 1998, respectively.

Evenness

In shelterwood, single-tree, and group selection stands, evenness was somewhat higher in 1994 than in the other 2 years, while values for 1996 and 1998 were almost identical (fig. 5). Avian evenness in clearcuts and controls followed different patterns. Consequently, differences among treatments were not significant in 1994 ($F = 1.53$; 4,80 df; $P = 0.2011$), 1996 ($F = 1.06$; 4,12 df; $P = 0.4197$), or 1998 ($F = 1.75$; 4,12 df; $P = 0.2029$).

CONCLUSIONS

Untreated, late-rotation stands occupying southerly aspects of the Ouachita Mountains seem to be characterized by low bird abundance, species richness, and diversity. However, our findings suggest that a variety of regeneration methods can be used to increase bird abundance, richness, and diversity through at least the fifth year after logging. The general pattern of these response variables suggests that they may be close to peak 5 years after treatment, but additional surveys are needed to substantiate this conclusion.

All stands were largely even-aged at study initiation. Consequently, stands that were selected for uneven-aged treatments are in a transition from an even- to an uneven-aged stand structure and will be for at least one or two additional harvest entries. Therefore, it is too early to draw definitive conclusions comparing uneven-aged treatments (single-tree

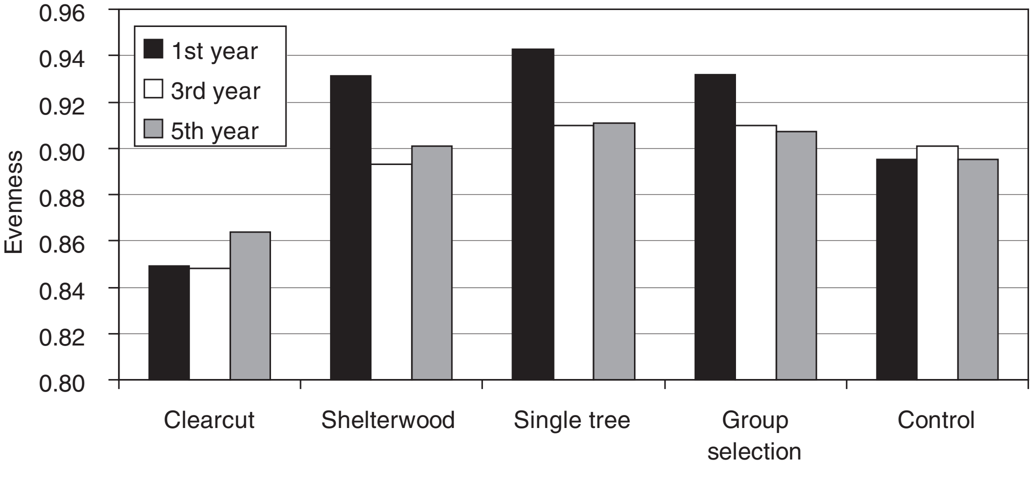

Figure 5—Bird evenness by silvicultural treatment during the first (1994), third (1996), and fifth years (1998) after harvesting of pine-hardwood stands in the Ouachita Mountains of Arkansas and Oklahoma. None of the within-year differences were significant ($P>0.05$).

and group selection) and even-aged (clearcut and shelterwood). Nevertheless, our data suggest that early-transition single-tree and group selection stands should have similar avian abundance, richness, and diversity as clearcut and shelterwood stands. Except for lower abundance in group selection stands during the fifth year, none of the other differences in response variables among these four regeneration methods were significant.

Due to space limitations, data on individual bird species responses to treatments are not presented here. However, it is important to note that the control stands, despite their lower avian abundance and diversity, supported some sensitive species that were uncommon or not encountered in harvested stands. Certain forest interior or disturbance-sensitive species, such as ovenbirds (*Seiurus aurocapillus*) and scarlet tanagers (*Piranga olivacea*), were relatively common in control stands but were absent or rare in harvested stands. Consequently, retention of mature stands is important for the conservation of these species. Furthermore, our preliminary findings suggest that a mix of silvicultural treatments over the landscape should enhance avian species diversity.

Young shelterwood stands may be aesthetically more pleasing than stands that are clearcut and planted (Personal communication. Victor A. Rudis. 2000. Research Forester. USDA Forest Service, Southern Research Station, Forestry Sciences Laboratory, 201 Lincoln Green, Starkville, MS 39759). Retention of seed trees in perpetuity should further increase public acceptance of shelterwood regeneration methods. Our findings indicate that stands regenerated using shelterwood methods on similar sites in the Ouachita Mountains should have comparable levels of avian abundance, richness, and diversity as clearcuts through at least the fifth year after harvest.

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EARLY SUCCESSION BIRD COMMUNITIES OF GROUP-SELECTION OPENINGS AND CLEARCUTS IN THE OUACHITA MOUNTAINS, ARKANSAS AND OKLAHOMA

Philip A. Tappe, Ronald E. Thill, David G. Peitz, and Roger W. Perry[1]

Abstract—We compared species presence/absence and density of nongame birds across a range of group-selection opening sizes and clearcuts. Group openings and clearcuts were harvested the summer of 1993. Birds were surveyed during 1995, 1996, and 1998 in 12 group openings ranging in size from 0.54 to 2.62 ac and within 4 clearcuts of approximately 35-40 ac each. Several species occurred throughout a variety of available opening sizes and many had their highest densities recorded in group openings. Bird densities were only related with opening size during 1995. No correlations with opening size were apparent for densities of individual species. Results indicate that group-selection cuts in late-rotation, shortleaf pine-dominated stands may help enhance relative breeding bird abundance. However, our data reflect bird responses to the initial transition of an uneven-aged forest structure, and may not adequately represent responses in the latter part of this transition. Additionally, data are needed on reproductive success of birds utilizing these openings under a variety of successional stages.

INTRODUCTION

Recent changes in management of national forests has increased interest in silvicultural alternatives to clearcutting and planting for the regeneration of pines in the southeastern United States. One alternative silvicultural method receiving increased use in southern national forests is group-selection management. Group-selection management consists of periodic harvesting of small groups of trees to create and maintain an uneven-aged stand structure with at least three distinct age/size classes. Cutting intervals on southern national forests generally range from 5 to 10 years for pine, with opening sizes from 0.5 to 2.0 ac (Kerpez 1994). If the stand surrounding these openings (termed the matrix) contains sufficient timber volume, the matrix is typically thinned at the same time these openings are created. Group-selection management is normally used to regenerate shade intolerant tree species. Little information exists on the effects of group selection harvesting on wildlife in southern pine and pine-hardwood forests.

Structural and floristic habitat alterations that result from group-selection harvesting may affect ecologically important faunal groups such as nongame birds. Nongame birds help facilitate seed and fungi dispersal, help control insect numbers, play essential roles in food web dynamics, and can create habitat for other wildlife species through excavation of cavities. In addition to their ecological values, nongame birds are important as a recreational resource to millions of people that watch and feed birds (U.S. Department of the Interior 1992). Neotropical migratory birds are of particular research interest due to recent evidence of long-term population declines in many species (Finch 1991).

Forest edges, such as those created by openings in forest canopies, are generally considered to be detrimental to many avian species because of possible increases in nest predation and parasitism by the brown-headed cowbird (*Molothrus ater*) (Brittingham and Temple 1983, Gates and Gysel 1978, Temple and Cary 1988, Wilcove 1985, Yahner and Scott 1988). Indices of habitat fragmentation, such as interior/edge ratios, fragment/matrix ratios, habitat patch heterogeneity, and the absolute amount of edge, are proportional to opening sizes (Rolstad 1991). Forest interior and area sensitive bird species may be adversely affected by management practices that increase forest fragmentation.

Few studies have assessed the effects of different opening sizes on nongame birds within forested landscapes. Lanham and Guynn (1998) found that species richness and relative abundance of neotropical migrants were positively related to increasing clearcut size, ranging from 1.24 to 32.12 ac, at piedmont sites in South Carolina. Using mist nets in bottomland hardwood forest gaps of 0.08, 0.31, and 1.24 ac, Kilgo and others (1999) captured more fall migrants in larger than smaller openings. However, the suitability of group-selection openings for breeding birds that utilize early succession habitats is unknown. Here, we compared presence/absence and densities of nongame birds among a range of group-selection opening sizes with those found in 35-40 ac clearcuts.

METHODS

Study Areas

We selected a subset of the 52 forest stands included in the Phase II (stand-level) Ecosystem Management Study described by Baker (1994). This subset consisted of eight stands located in Arkansas and Oklahoma on the Ouachita National Forest and the southern-most district of the Ozark/St. Francis National Forest. All stands were late-rotation (60-70 years old), 35-40 ac shortleaf pine- (*Pinus echinata*

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Mill.) dominated stands with mostly south, southeastern, or southwestern aspects. Prior to harvest, coniferous basal area of the 52 Phase II stands averaged 95.3 ft^2/ac; hardwood basal area averaged 34.3 ft^2/ac (Guldin and others 1994). Pretreatment stand and habitat conditions are described by Guldin and others (1994) and Thill and others (1994).

Treatments

Although the overall Phase II study included 13 treatments, this paper contrasts bird communities within group openings of different size in the pine/hardwood group-selection treatment and the clearcut treatment. All harvesting was completed between June 1 and September 30, 1993, and site preparation was completed between October 1, 1993 and April 30, 1994.

Group-selection stands—Approximately 10 percent of each of the four stands receiving the pine/hardwood group-selection treatment was harvested, creating a total of twelve group openings with the following sizes: 0.54, 0.59, 0.60, 0.72, 0.75, 0.92, 0.95, 0.98, 1.12, 1.23, 1.48, and 2.62 ac. All pines, except a few smaller trees totaling an average of only 1.3 ft^2/ac basal area, were removed from the openings. Hardwoods < 5.9 in. d.b.h. were felled and left on site. A residual overstory hardwood component averaging 17.0 ft^2/ac basal area was retained in the openings to improve aesthetics, forest diversity, and hard mast availability for wildlife. Pines in the matrix surrounding group openings were thinned to 70 to 80 ft^2/ac; no other management occurred within the matrices.

Clearcuts—All four stands receiving the clearcut treatment contained ephemeral drainages that typically flow only for brief periods during high rainfall events. Unharvested linear strips ("greenbelts") were retained along these drainages for watershed protection. Averaging about 11 percent per stand, greenbelts were comprised of woody vegetation that was similar in composition and structure to pretreatment stand conditions. Except for these greenbelts and a few overstory hardwoods (averaging 2-5 ft^2/ac basal area), all merchantable pines and hardwoods were harvested from the four clearcut stands. All remaining trees were injected with herbicide (Baker 1994). Although clearcuts were scheduled to be ripped prior to planting, contractors could not be found to rip two of the stands. All stands were planted with genetically improved pines on an 8- by 10-ft spacing between December 1994 and March 1995. One stand was replanted in February 1996 due to inadequate stocking.

Bird Surveys

Relative bird abundance was quantified within a 131-ft-radius (40 m) circular plot centered in each of the 12 group openings. Additionally, one plot per clearcut stand was randomly selected from the five plots used for the Phase II Ecosystem Management bird surveys (Thill and others, in press). Plots were surveyed six times, once by six different observers, for 10 minutes. All plots were surveyed within 3.5 hours of sunrise. Surveys were conducted during May and June of 1995, 1996, and 1998. Birds were recorded as either in or outside of the 131-ft-radius plots. Birds within the plots were recorded by species, sex (if known), and singing status; birds outside the plots were recorded as present only. Since a 131-ft-radius plot encompasses an area of 1.24 ac, larger than most group openings studied, many group-opening plots included both opening and matrix habitat. Thus, birds were also recorded as either in the opening or in the surrounding thinned matrix.

Analyses

Because sampling plots within group-selection stands often contained both opening and matrix habitat, bird data for the openings only were converted to numbers of birds per ac; i.e., density values. The mean number of birds encountered by all observers on each group-opening plot was computed using birds recorded within the actual opening only, and for each clearcut plot. Clearcut plots did not include any greenbelt inclusions, thus the area sampled was 1.24 ac per clearcut.

Densities were computed for each opening size for three sets of birds: (1) all birds combined (males and females), (2) migrants, and (3) residents. Pearson's correlation coefficients were computed to test ($\alpha \leq 0.05$) for linear relationships between group-opening size (including the four clearcuts) and mean total density for all birds combined, migrants, and residents. Correlation coefficients were also used to test ($\alpha \leq 0.05$) for relationships between group-opening size and densities of individual species.

RESULTS

A total of 1,383 breeding birds representing 53 species were recorded over all years (tables 1 and 6). Presence/absence and density information by opening size are summarized in tabular form (tables 1, 2, 3, and 4). Four species comprised almost 50 percent of the individuals recorded: indigo bunting, Carolina wren, prairie warbler, and yellow-breasted chat. Seventeen percent of species were recorded only in clearcuts, 20.7 percent were recorded only in group-selection openings, and 62.3 percent were recorded in both clearcuts and group openings (table 1). The indigo bunting was the only species to occur in all group-opening sizes and the clearcuts. Over all years, the Carolina wren occurred in all but the 0.60 ac opening, the summer tanager occurred in all but the 0.60 and 0.72 ac openings, and the black-and-white warbler occurred in all but the 0.60 and 0.75 ac openings. All other species were absent in ≥ 3 openings. However, species presence or absence by opening size varied by year (tables 2, 3, and 4). Over all years, 37 species (69.8 percent) had highest average densities recorded in a group opening; 13 species (24.5 percent) had their highest densities in the clearcut treatment (table 1). Of the 33 species found in both group openings and clearcuts, only 4 had their highest average densities in clearcuts: the prairie warbler, brown-headed cowbird, common yellowthroat, and eastern bluebird. However, density by opening size also varied by year (tables 2, 3, and 4).

A significant, positive linear relationship between opening size and mean total bird density was found during 1995 ($r = 0.68$, $P = 0.004$). However, no linear relationships were evident during 1996 or 1998 (table 5 and fig. 1). Furthermore, no densities of individual species were correlated with opening size ($P > 0.05$).

Table 1—Densities of birds (birds per ac), averaged across birders and years (1995, 1996, and 1998), during point counts in 12 group openings (0.54-2.62 ac) and 4 clearcuts (35-40 ac) during May-June in the Ouachita Mountains of Arkansas and Oklahoma

			Opening size												Clearcuts
Species	Density	Total	0.54	0.59	0.60	0.72	0.75	0.92	0.95	0.98	1.12	1.23	1.48	2.62	35–40
	no.	*percent*	*acres*												
Indigo bunting*	408	29.5	1.5	2.1	2.5	1.7	1.9	1.6	1.7	1.1	1.0	1.3	0.8	1.0	1.5
Carolina wren	101	7.3	1.2	0.5	—	0.3	0.4	0.4	0.2	0.3	0.3	0.4	0.2	0.5	0.3
Prairie warbler*	91	6.6	—	0.2	—	—	0.2	0.5	—	0.2	0.1	0.1	0.1	—	0.8
Yellow-breasted chat*	90	6.5	—	—	0.3	—	0.8	0.2	—	—	0.3	0.5	—	0.2	0.6
Brown-headed cowbird	66	4.8	0.1	0.2	—	—	0.2	0.1	0.1	0.1	0.1	0.2	—	0.2	0.5
Summer tanager*	64	4.6	0.2	0.3	—	—	0.5	0.5	0.1	0.1	0.3	0.5	0.1	0.1	0.2
Mourning dove	38	2.7	—	—	—	—	—	—	—	—	—	—	—	—	0.4
American goldfinch	32	2.3	—	—	0.3	—	—	—	0.4	—	—	—	—	0.1	0.2
Carolina chickadee	30	2.2	0.2	0.1	0.4	—	0.3	—	0.1	—	0.2	0.1	0.2	0.2	< 0.1
Pine warbler	30	2.2	—	0.4	0.2	0.2	0.2	0.2	—	0.2	0.2	—	0.5	—	< 0.1
Blue-gray gnatcatcher*	29	2.1	0.2	—	—	—	0.7	0.1	0.1	—	0.2	0.2	—	0.1	0.1
Great-crested flycatcher*	28	2.0	—	—	—	—	0.2	—	0.1	0.1	—	0.1	—	0.1	0.2
Eastern wood pewee*	27	2.0	0.4	—	—	—	0.2	—	0.4	—	0.1	0.1	—	0.2	0.1
Common yellowthroat*	24	1.7	—	—	0.1	—	0.1	—	—	—	—	0.1	—	0.1	0.2
Black & white warbler*	24	1.7	0.2	0.1	—	0.1	—	0.1	0.1	0.1	0.1	0.2	0.1	0.1	0.1
Red-headed woodpecker	24	1.7	—	—	—	—	—	—	—	—	—	—	—	—	0.3
Ruby-throated hummingbird*	21	1.5	0.3	—	—	0.2	0.3	—	0.2	—	0.1	0.2	—	0.1	< 0.1
White-eyed vireo*	19	1.4	0.1	—	—	—	0.4	—	—	—	0.3	0.1	—	< 0.1	0.1
Red-eyed vireo*	19	1.4	—	0.1	—	—	0.3	—	—	0.1	0.1	0.1	0.1	0.1	< 0.1
Tufted-titmouse	19	1.4	0.1	—	—	—	0.2	0.1	—	—	—	0.1	—	0.5	< 0.1
Eastern bluebird	18	1.3	—	—	0.1	—	—	—	—	—	—	—	—	—	0.2
White-breasted nuthatch	17	1.2	0.4	—	—	—	—	—	0.1	—	0.1	0.1	—	0.1	0.1
Field sparrow	16	1.2	—	—	—	0.2	—	0.1	—	—	—	—	—	—	0.1
Northern cardinal	15	1.1	—	—	—	—	—	—	—	—	0.3	0.4	—	—	< 0.1
Bachman’s sparrow*	14	1.0	—	—	—	—	—	0.1	—	0.1	—	—	—	—	0.1
Eastern kingbird*	12	0.9	—	—	—	—	—	—	—	—	—	—	—	—	0.1
Bluejay	10	0.7	—	—	—	—	0.2	—	—	—	—	0.1	—	—	0.1
Common flicker	7	0.5	—	—	—	—	—	—	—	—	—	—	—	—	0.1
Lark sparrow*	7	0.5	—	—	—	—	—	—	—	—	—	—	—	—	0.1
Hooded warbler*	6	0.4	—	—	—	—	—	—	—	—	—	0.1	—	0.1	< 0.1
Downy woodpecker	5	0.4	—	—	—	—	0.1	—	—	—	—	—	—	0.1	< 0.1
Hairy woodpecker	5	0.4	0.1	—	0.1	—	—	—	0.1	—	—	0.1	< 0.1	—	—
Scarlet tanager*	5	0.4	—	—	—	—	—	—	—	—	—	0.1	—	—	< 0.1
Pileated woodpecker	4	0.3	—	—	0.1	—	—	0.1	—	—	—	0.1	—	< 0.1	—
Red-bellied woodpecker	4	0.3	0.1	—	—	—	—	—	—	0.1	0.1	0.1	—	—	—
Yellow-throated vireo*	4	0.3	—	—	—	—	—	0.1	—	—	—	—	—	< 0.1	< 0.1
Baltimore oriole*	4	0.3	—	—	—	—	0.1	—	—	—	—	—	—	< 0.1	< 0.1

(continued)

Table 1—Densities of birds (birds per ac), averaged across birders and years (1995, 1996, and 1998), during point counts in 12 group openings (0.54-2.62 ac) and 4 clearcuts (35-40 ac) during May-June in the Ouachita Mountains of Arkansas and Oklahoma (continued)

			Opening size												Clearcuts
Species	Density	Total	0.54	0.59	0.60	0.72	0.75	0.92	0.95	0.98	1.12	1.23	1.48	2.62	35–40
	no.	*percent*	- - - - -	- - - - -	- - - - -	- - - - -	- - - - -	- - - - -	*acres*	- - - - -	- - - - -	- - - - -	- - - - -	- - - - -	- - - - -
American kestrel	4	0.3	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Chipping sparrow	3	0.2	—	—	—	—	0.1	—	—	—	—	—	—	—	< 0.1
Broad-winged hawk*	3	0.2	0.1	—	—	—	—	—	—	—	—	—	< 0.1	< 0.1	—
Orchard oriole*	3	0.2	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Eastern phoebe	3	0.2	—	—	—	—	—	0.1	—	—	—	—	—	—	< 0.1
Yellow-billed cuckoo*	2	0.1	0.2	—	—	—	—	—	—	—	—	—	—	—	—
Brown-headed nuthatch	2	0.1	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Yellow-throated warbler*	2	0.1	—	—	—	—	—	—	0.1	—	—	—	—	—	< 0.1
N. rough-winged swallow*	2	0.1	—	—	—	—	0.2	—	—	—	—	—	—	—	—
Worm-eating warbler*	1	0.1	—	—	—	—	—	—	—	—	—	—	—	< 0.1	—
American crow	1	0.1	—	—	—	—	—	—	0.1	—	—	—	—	—	—
Brown thrasher	1	0.1	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Gray catbird*	1	0.1	—	—	—	—	—	0.1	—	—	—	—	—	—	—
Ovenbird*	1	0.1	—	—	—	—	—	—	—	—	—	0.1	—	—	—
Total	1,383	100.0	5.4	4.1	4.2	2.7	7.8	4.4	3.9	2.5	4.0	5.8	2.2	4.0	7.2

Asterisks after bird names denote migrant species.

Table 2—Densities of birds (birds per ac), averaged across birders, during point counts in 12 group openings (0.54-2.62 ac) and 4 clearcuts (35-40 ac) during May-June 1995 in the Ouachita Mountains of Arkansas and Oklahoma

			Opening size												Clearcuts
Species	Density	Total	0.54	0.59	0.60	0.72	0.75	0.92	0.95	0.98	1.12	1.23	1.48	2.62	35–40
	no.	*percent*						*acres*							
Indigo bunting*	186	29.6	1.2	1.4	2.5	1.6	2.9	1.8	2.5	1.7	1.3	1.8	1.1	1.6	2.4
Carolina wren	71	11.3	3.1	0.9	—	0.9	0.7	—	0.4	0.9	0.7	0.7	0.5	1.2	0.7
Prairie warbler*	21	3.3	—	0.3	—	—	—	—	—	—	—	—	—	—	0.7
Yellow-breasted chat*	26	4.1	—	—	—	—	0.2	—	—	—	—	—	—	—	0.8
Brown-headed cowbird	35	5.5	—	0.6	—	—	—	0.2	—	—	0.3	0.4	—	—	0.9
Summer tanager*	32	5.1	—	—	—	—	1.1	0.7	0.2	0.3	0.7	0.7	0.3	0.3	0.2
Mourning dove	29	4.6	—	—	—	—	—	—	—	—	—	—	—	—	1.0
American goldfinch	22	3.5	—	—	0.8	—	—	—	0.2	—	—	—	—	0.3	0.5
Carolina chickadee	1	0.2	—	0.3	—	—	—	—	—	—	—	—	—	—	—
Pine warbler	9	1.4	—	0.3	0.6	0.2	—	—	—	—	0.3	—	0.3	—	< 0.1
Blue-gray gnatcatcher*	16	2.5	0.6	—	—	—	1.1	0.2	0.2	—	—	0.5	—	0.3	< 0.1
Great-crested flycatcher*	14	2.2	—	—	—	—	—	—	0.2	—	—	0.3	—	—	0.4
Eastern wood pewee*	12	1.9	0.6	—	—	—	0.4	—	0.5	—	0.2	0.1	—	—	0.1
Common yellowthroat*	10	1.6	—	—	—	—	—	—	—	—	—	—	—	0.3	0.3
Black & white warbler*	9	1.4	—	0.3	—	0.2	—	0.2	—	—	—	0.3	0.1	0.1	0.1
Red-headed woodpecker	19	3.0	—	—	—	—	—	—	—	—	—	—	—	—	0.6
Ruby-throated hummingbird*	4	0.6	—	—	—	0.2	0.2	—	—	—	—	0.3	—	—	—
White-eyed vireo*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Red-eyed vireo*	7	1.1	—	—	—	—	0.4	—	—	—	0.3	0.1	0.1	0.1	—
Tufted-titmouse	8	1.3	—	—	—	—	—	—	—	—	—	—	—	0.8	< 0.1
Eastern bluebird	13	2.1	—	—	0.3	—	—	—	—	—	—	—	—	—	0.4
White-breasted nuthatch	6	1.0	—	—	—	—	—	—	0.2	—	0.2	—	—	—	0.1
Field sparrow	9	1.4	—	—	—	—	—	—	—	—	—	—	—	—	0.3
Northern cardinal	10	1.6	—	—	—	—	—	—	—	—	0.5	0.7	—	—	0.1
Bachman's sparrow*	14	2.2	—	—	—	—	—	0.2	—	0.2	—	—	—	—	0.4
Eastern kingbird*	2	0.3	—	—	—	—	—	—	—	—	—	—	—	—	0.1
Bluejay	8	1.3	—	—	—	—	—	—	—	—	—	0.4	—	—	0.2
Kentucky warbler*	1	0.2	—	—	—	—	—	—	—	—	—	—	—	0.1	—
Blue grosbeak*	1	0.2	—	—	0.3	—	—	—	—	—	—	—	—	—	—
Common flicker	1	0.2	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Lark sparrow*	6	1.0	—	—	—	—	—	—	—	—	—	—	—	—	0.2
Hooded warbler*	2	0.3	—	—	—	—	—	—	—	—	—	—	—	—	0.1
Downy woodpecker	3	0.5	—	—	—	—	0.2	—	—	—	—	—	—	0.1	< 0.1
Hairy woodpecker	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Scarlet tanager*	2	0.3	—	—	—	—	—	—	—	—	—	0.1	—	—	< 0.1
Pileated woodpecker	3	0.5	—	—	—	—	—	0.2	—	—	—	0.1	—	0.1	—
Red-bellied woodpecker	1	0.2	—	—	—	—	—	—	0.2	—	—	—	—	—	—

(continued)

Table 2—Densities of birds (birds per ac), averaged across birders, during point counts in 12 group openings (0.54-2.62 ac) and 4 clearcuts (35-40 ac) during May-June 1995 in the Ouachita Mountains of Arkansas and Oklahoma

			Opening size											Clearcuts
Species	Density	Total	0.54	0.59	0.60	0.72	0.75	0.92	0.95	0.98	1.12	1.23	1.48	2.62	35–40
	no.	*percent*	*acres*											
Yellow-throated vireo*	2	0.3	—	—	—	—	—	—	—	—	—	—	—	0.1	< 0.1
Baltimore oriole*	1	0.2	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
American kestrel	2	0.3	—	—	—	—	—	—	—	—	—	—	—	—	0.1
Chipping sparrow	1	0.2	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Broad-winged hawk*	1	0.2	—	—	—	—	—	—	—	—	—	—	0.1	—	—
Orchard oriole*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Eastern phoebe	3	0.3	—	—	—	—	—	0.2	—	—	—	—	—	—	0.1
Yellow-billed cuckoo*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Brown-headed nuthatch	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Yellow-throated warbler*	2	0.3	—	—	—	—	—	—	0.2	—	—	—	—	—	< 0.1
N. rough-winged swallow*	2	0.3	—	—	—	—	0.4	—	—	—	—	—	—	—	—
Worm-eating warbler*	1	0.2	—	—	—	—	—	—	—	—	—	—	—	0.1	—
American crow	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Brown thrasher	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Gray catbird*	1	0.2	—	—	—	—	—	0.2	—	—	—	—	—	—	—
Ovenbird*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Total	629	100.0	5.5	4.1	4.5	3.1	7.6	3.9	4.8	3.1	4.5	6.5	2.5	5.5	11.2

Asterisks after bird names denote migrant species.

Table 3—Densities of birds (birds per ac), averaged across birders, identified during point counts in 12 group openings (0.54-2.62 ac) and 4 clearcuts (35-40 ac) during May-June, 1996 in the Ouachita Mountains of Arkansas and Oklahoma

			Opening size												Clearcuts
Species	Density	Total	0.54	0.59	0.60	0.72	0.75	0.92	0.95	0.98	1.12	1.23	1.48	2.62	35–40
	no.	*percent*	*acres*												
Indigo bunting*	106	29.3	1.5	1.7	3.1	2.1	1.8	1.5	1.2	0.5	0.9	1.5	0.3	1.1	0.7
Carolina wren	16	4.4	0.6	—	—	—	—	0.7	0.2	—	0.2	0.4	—	0.3	0.1
Prairie warbler*	30	8.3	—	—	—	—	0.7	0.9	—	—	0.2	0.1	—	—	0.7
Yellow-breasted chat*	32	8.9	—	—	—	—	0.9	0.5	—	—	0.3	0.8	—	0.5	0.4
Brown-headed cowbird	17	4.7	—	—	—	—	—	0.2	0.4	0.2	—	0.3	—	0.3	0.3
Summer tanager*	20	5.5	0.6	0.3	—	—	0.4	0.5	0.2	—	—	0.7	—	—	0.2
Mourning dove	5	1.4	—	—	—	—	—	—	—	—	—	—	—	—	0.2
American goldfinch	7	1.9	—	—	—	—	—	—	0.5	—	—	—	—	0.1	0.1
Carolina chickadee	10	2.7	—	—	0.6	—	—	—	—	—	0.5	—	—	0.5	< 0.1
Pine warbler	9	2.5	—	—	—	—	0.4	0.5	—	0.3	0.2	—	0.1	—	—
Blue-gray gnatcatcher*	5	1.4	—	—	—	—	0.2	—	—	—	0.3	—	—	—	0.1
Great-crested flycatcher*	9	2.5	—	—	—	—	0.4	—	—	—	—	—	—	—	0.2
Eastern wood pewee*	11	3.0	0.6	—	—	—	0.2	—	0.4	—	—	0.1	—	0.4	0.1
Common yellowthroat*	9	2.5	—	—	—	—	0.2	—	—	—	—	0.4	—	—	0.2
Black & white warbler*	5	1.4	—	—	—	—	—	0.2	0.2	0.2	—	—	—	0.1	< 0.1
Red-headed woodpecker	4	1.1	—	—	—	—	—	—	—	—	—	—	—	—	0.1
Ruby-throated hummingbird*	10	2.7	0.9	—	—	0.2	—	—	0.4	—	0.3	0.1	—	—	< 0.1
White-eyed vireo*	1	0.3	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Red-eyed vireo*	5	1.4	—	—	—	—	0.2	—	—	0.2	—	0.1	0.1	0.1	—
Tufted-titmouse	4	1.1	—	—	—	—	—	0.2	—	—	—	0.1	—	0.3	—
Eastern bluebird	3	0.8	—	—	—	—	—	—	—	—	—	—	—	—	0.1
White-breasted nuthatch	8	2.2	1.2	—	—	—	—	—	—	—	—	0.3	—	—	0.1
Field sparrow	3	0.8	—	—	—	—	—	—	—	—	—	—	—	—	0.1
Northern cardinal	2	0.6	—	—	—	—	—	—	—	—	—	0.3	—	—	—
Bachman’s sparrow*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Eastern kingbird*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Bluejay	2	0.6	—	—	—	—	0.4	—	—	—	—	—	—	—	—
Kentucky warbler*	1	0.3	—	—	—	—	—	—	—	—	—	0.1	—	—	—
Blue grosbeak*	4	1.1	—	—	—	—	—	—	—	—	—	—	—	—	0.1
Common flicker	5	1.4	—	—	—	—	—	—	—	—	—	—	—	—	0.2
Lark sparrow*	1	0.3	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Hooded warbler*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Downy woodpecker	1	0.3	—	—	—	—	—	—	—	—	—	—	—	0.1	—
Hairy woodpecker	3	0.8	0.3	—	—	—	—	—	0.2	—	—	0.1	—	—	—
Scarlet tanager*	2	0.6	—	—	—	—	—	—	—	—	—	—	—	—	0.1
Pileated woodpecker	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Red-bellied woodpecker	3	0.8	0.3	—	—	—	—	—	—	—	0.2	0.1	—	—	—

(continued)

Table 3—Densities of birds (birds per ac), averaged across birders, identified during point counts in 12 group openings (0.54-2.62 ac) and 4 clearcuts (35-40 ac) during May-June, 1996 in the Ouachita Mountains of Arkansas and Oklahoma

			Opening size												Clearcuts
Species	Density	Total	0.54	0.59	0.60	0.72	0.75	0.92	0.95	0.98	1.12	1.23	1.48	2.62	35–40
	no.	*percent*	acres												
Yellow-throated vireo*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Baltimore oriole*	1	0.3	—	—	—	—	—	—	—	—	—	—	—	0.1	—
American kestrel	2	0.6	—	—	—	—	—	—	—	—	—	—	—	—	0.1
Chipping sparrow	1	0.3	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Broad-winged hawk*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Orchard oriole*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Eastern phoebe	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Yellow-billed cuckoo*	2	0.6	0.6	—	—	—	—	—	—	—	—	—	—	—	—
Brown-headed nuthatch	2	0.6	—	—	—	—	—	—	—	—	—	—	—	—	0.1
Yellow-throated warbler*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
N. rough-winged swallow*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Worm-eating warbler*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
American crow	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Brown thrasher	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Gray catbird*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Ovenbird*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Total	361	100.0	6.6	2.0	3.7	2.3	5.8	5.2	3.7	1.4	3.1	5.5	0.5	3.9	4.5

Asterisks after bird names denote migrant species.

Table 4—Densities of birds (birds per ac), averaged across birders, during point counts in 12 group openings (0.54-2.62 ac) and 4 clearcuts (35-40 ac) during May-June, 1998 in the Ouachita Mountains of Arkansas and Oklahoma

			Opening size												Clearcuts
Species	Density	Total	0.54	0.59	0.60	0.72	0.75	0.92	0.95	0.98	1.12	1.23	1.48	2.62	35–40
	no.	*percent*	*acres*												
Indigo bunting*	116	29.5	1.8	3.1	1.9	1.4	1.1	1.5	1.4	1.0	0.9	0.5	1.1	0.3	1.3
Carolina wren	14	3.6	—	0.6	—	—	0.4	0.4	—	—	—	—	—	0.1	0.2
Prairie warbler*	40	10.1	—	0.3	—	—	—	0.5	—	0.7	—	—	0.4	—	1.0
Yellow-breasted chat*	32	8.1	—	—	0.8	—	1.3	—	—	—	0.5	0.7	—	—	0.5
Brown-headed cowbird	14	3.6	0.3	—	—	—	0.7	—	—	—	—	—	—	0.3	0.3
Summer tanager*	12	3.1	—	0.6	—	—	—	0.4	—	—	0.2	0.1	0.1	0.1	0.1
Mourning dove	4	1.0	—	—	—	—	—	—	—	—	—	—	—	—	0.1
American goldfinch	3	0.8	—	—	—	—	—	—	0.4	—	—	—	—	—	< 0.1
Carolina chickadee	19	4.8	0.6	0.6	—	—	0.9	—	0.4	—	—	0.3	0.7	0.1	< 0.1
Pine warbler	12	3.1	—	0.9	—	0.2	—	—	—	0.2	—	—	0.9	—	—
Blue-gray gnatcatcher*	8	2.0	—	—	—	—	0.7	—	—	—	0.2	0.1	—	—	0.1
Great-crested flycatcher*	5	1.3	—	—	—	—	—	—	—	0.2	—	—	—	0.3	0.1
Eastern wood pewee*	4	1.0	—	—	—	—	—	—	0.2	—	—	—	—	0.3	< 0.1
Common yellowthroat*	5	1.3	—	—	0.3	—	—	—	—	—	—	—	—	—	0.1
Black & white warbler*	10	2.5	0.6	—	—	—	—	—	—	—	0.2	0.3	0.3	0.1	0.1
Red-headed woodpecker	1	0.3	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Ruby-throated hummingbird*	7	1.8	—	—	—	—	0.7	—	0.2	—	—	0.1	—	0.3	—
White-eyed vireo*	18	4.6	0.3	—	—	—	1.3	—	—	—	0.7	0.1	—	0.1	0.1
Red-eyed vireo*	7	1.8	—	0.3	—	—	0.2	—	—	0.2	—	0.1	0.1	0.1	< 0.1
Tufted-titmouse	7	1.8	0.3	—	—	—	0.4	—	—	—	—	—	—	0.5	—
Eastern bluebird	2	0.4	—	—	—	—	—	—	—	—	—	—	—	—	0.1
White-breasted nuthatch	3	0.8	—	—	—	—	—	—	—	—	—	—	—	0.3	< 0.1
Field sparrow	4	1.0	—	—	—	0.5	—	0.2	—	—	—	—	—	—	< 0.1
Northern cardinal	3	0.8	—	—	—	—	—	—	—	—	0.3	0.1	—	—	—
Bachman's sparrow*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Eastern kingbird*	10	2.5	—	—	—	—	—	—	—	—	—	—	—	—	0.3
Bluejay	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Kentucky warbler*	7	1.8	—	—	—	—	0.4	—	—	—	0.3	0.4	—	—	—
Blue grosbeak*	3	0.8	—	0.3	—	—	—	—	—	—	—	0.1	—	—	< 0.1
Common flicker	1	0.3	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Lark sparrow*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Hooded warbler*	4	1.0	—	—	—	—	—	—	—	—	—	0.1	—	0.4	—
Downy woodpecker	1	0.3	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Hairy woodpecker	2	0.4	—	—	0.3	—	—	—	—	—	—	—	0.1	—	—
Scarlet tanager*	1	0.3	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Pileated woodpecker	1	0.3	—	—	0.3	—	—	—	—	—	—	—	—	—	—
Red-bellied woodpecker	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—

(continued)

Table 4—Densities of birds (birds per ac), averaged across birders, during point counts in 12 group openings (0.54-2.62 ac) and 4 clearcuts (35-40 ac) during May-June, 1998 in the Ouachita Mountains of Arkansas and Oklahoma

			Opening size												Clearcuts
Species	Density	Total	0.54	0.59	0.60	0.72	0.75	0.92	0.95	0.98	1.12	1.23	1.48	2.62	35–40
	no.	*percent*	*acres*												
Yellow-throated vireo*	2	0.4	—	—	—	—	—	0.2	—	—	—	—	—	—	< 0.1
Baltimore oriole*	2	0.4	—	—	—	—	0.2	—	—	—	—	—	—	—	< 0.1
American kestrel	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Chipping sparrow	1	0.3	—	—	—	—	0.2	—	—	—	—	—	—	—	—
Broad-winged hawk*	2	0.4	0.3	—	—	—	—	—	—	—	—	—	—	0.1	—
Orchard oriole*	3	0.8	—	—	—	—	—	—	—	—	—	—	—	—	0.1
Eastern phoebe	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Yellow-billed cuckoo*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Brown-headed nuthatch	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Yellow-throated warbler*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
N. rough-winged swallow*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Worm-eating warbler*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
American crow	1	0.3	—	—	—	—	—	—	0.2	—	—	—	—	—	—
Brown thrasher	1	0.3	—	—	—	—	—	—	—	—	—	—	—	—	< 0.1
Gray catbird*	0	0	—	—	—	—	—	—	—	—	—	—	—	—	—
Ovenbird*	1	0.3	—	—	—	—	—	—	—	—	—	0.1	—	—	—
Total	393	100.0	4.2	6.7	3.6	2.1	8.5	3.2	2.8	2.3	3.3	3.1	3.7	3.4	5.0

Asterisks after bird names denote migrant species.

Table 5—Correlation coefficients of mean densities (birds per ac) with opening size (0.54-40 ac) during May-June, 1995, 1996 and 1998 in the Ouachita Mountains of Arkansas and Oklahoma

	1995		1996		1998	
Variable	*r*	*P*	*r*	*P*	*r*	*P*
Migrants	0.54	0.031	0.06	0.814	0.38	0.144
Residents	0.70	0.003	0.32	0.224	-0.05	0.868
Combined	0.68	0.004	0.22	0.419	0.31	0.242

Table 6—Common and scientific names of birds identified during point counts in12 group openings (0.54-2.62 ac) and 4 clearcuts (35-40 ac) during May-June, 1995-1996 and1998 in the Ouachita Mountains of Arkansas and Oklahoma

Species	Scientific name	Species	Scientific name
Indigo bunting*	*Passerina cyanea*	Bluejay	Cyanocitta cristata
Carolina wren	*Thryothorus ludovicianus*	Kentucky warbler*	*Oporomis formosus*
Prairie warbler*	*Dendroica discolor*	Blue grosbeak*	*Guiraca caerulea*
Yellow-breasted chat*	*Icteria virens*	Common flicker	*Colaptes auratus*
Brown-headed cowbird	*Molothrus ater*	Lark sparrow*	*Chondestes grammacus*
Summer tanager*	*Piranga rubra*	Hooded warbler*	*Wilsonia critrina*
Mourning dove	*Zenaida macroura*	Downy woodpecker	*Picoides pubescens*
American goldfinch	*Carduelis tristis*	Hairy woodpecker	*Picoides villosus*
Carolina chickadee	*Parus carolinensis*	Scarlet tanager*	*Piranga olivacea*
Pine warbler	*Dendroica pinus*	Pileated woodpecker	*Dryocopus pileatus*
Blue-gray gnatcatcher*	*Polioptila caerulea*	Red-bellied woodpecker	*Melanerpes carolinus*
Great-crested flycatcher*	*Myiarchus crinitus*	Yellow-throated vireo*	*Vireo flavifrons*
Eastern wood pewee*	*Contopus virens*	Baltimore oriole*	*Icterus galbula*
Common yellowthroat*	*Geothlypis trichas*	American kestrel	*Falco sparverius*
Black & white warbler*	*Mniotilta varia*	Chipping sparrow	Spizella paserina
Red-headed woodpecker	*Melanerpes erythrocephalus*	Broad-winged hawk*	*Buteo platypterus*
		Orchard oriole*	*Icterus spurius*
Ruby-throated hummingbird*	*Archilochus colubris*	Eastern phoebe	*Sayomis phoebe*
White-eyed vireo*	*Vireo griseus*	Yellow-billed cuckoo*	*Coccyzus americanus*
Red-eyed vireo*	*Vireo olivaceus*	Brown-headed nuthatch	*Sitta pusilla*
Tufted-titmouse	*Parus bicolor*	Yellow-throated warbler*	*Dendroica dominica*
Eastern bluebird	*Sialia sialis*	N. rough-winged swallow*	*Stelgidopteryx serripennis*
White-breasted nuthatch	*Sitta carolinensis*	Worm-eating warbler*	*Helmitheros vermivorus*
Field sparrow	*Spizella pusilla*	American crow	*Corvus brachyrhynchos*
Northern cardinal	*Cardinalis cardinalis*	Brown thrasher	*Toxostoma rufum*
Bachman's sparrow*	*Aimophila aestivalis*	Gray catbird*	*Dumetella carolinensis*
Eastern kingbird*	*Tyranus tyranus*	Ovenbird*	*Seiurus aurocapillus*

Asterisks after bird names denote migrant species.
Source: Hamel (1992).

Migrants

A total of 928 migrants of 29 species were recorded over all years (table 1). Migrants represented 54.7 percent of species recorded. Four species comprised over 70 percent of the individual migrants recorded: indigo bunting, prairie warbler, yellow-breasted chat, and summer tanager.

A weak but significant, positive linear relationship between opening size and mean total migrant density was found during 1995 (*r* = 0.54, *P* = 0.031). However, no linear relationships were evident during 1996 or 1998 (table 5 and fig. 1).

Residents

A total of 455 residents of 24 species were recorded over all years (table 1). Residents represented 45.3 percent of species recorded. Seven species comprised over 70 percent of the individual residents recorded: Carolina wren, brown-headed cowbird, mourning dove, American goldfinch, Carolina chickadee, pine warbler, and red-headed woodpecker.

A significant, positive linear relationship between opening size and mean total resident density was found during 1995

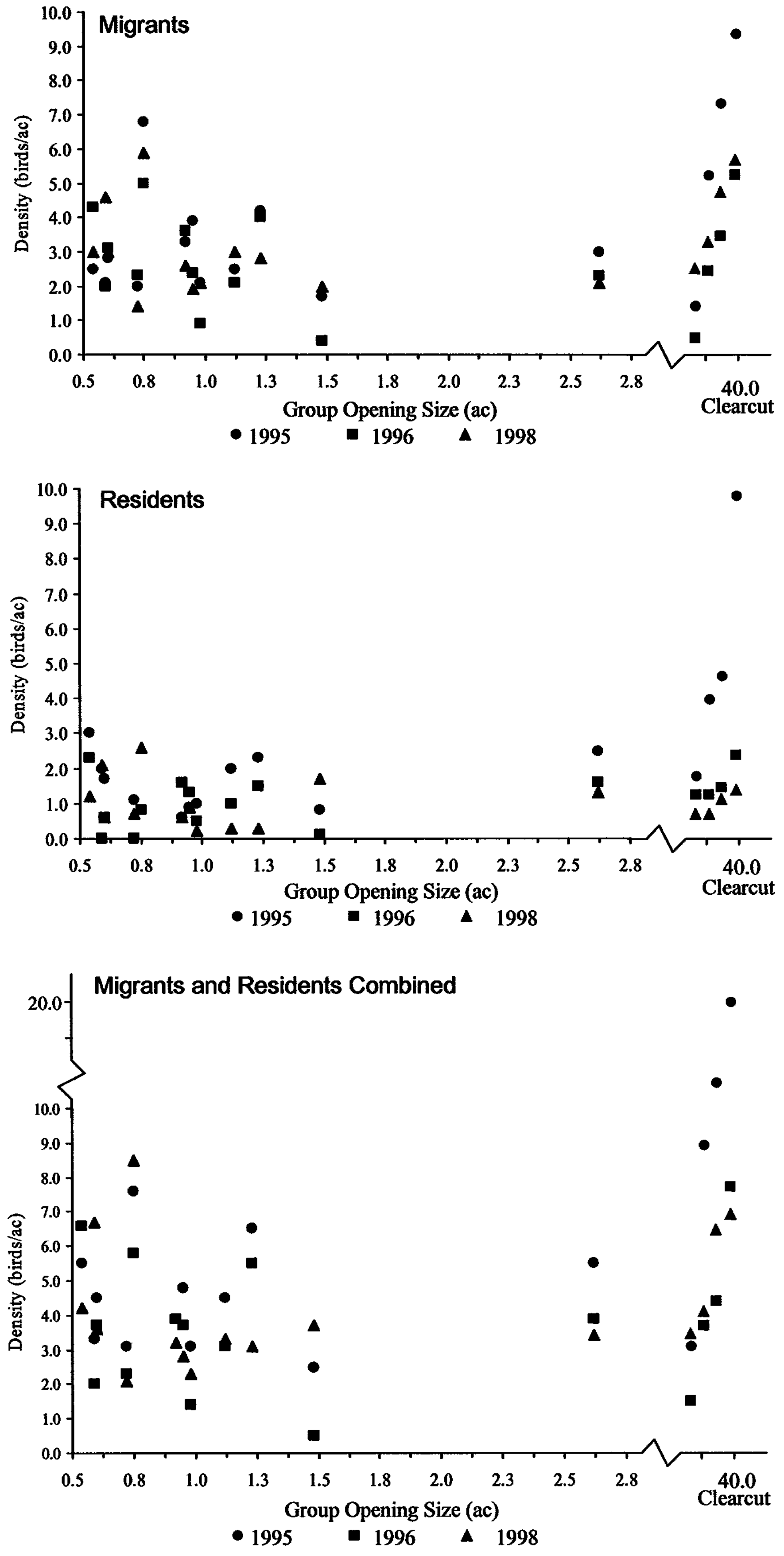

Figure 1—Densities of birds, averaged across birders, during point counts in 12 group openings (0.54-2.62 ac) and 4 clearcuts (35-40 ac) during May-June, 1995, 1996, and 1998 in the Ouachita Mountains of Arkansas and Oklahoma.

($r = 0.70$, $P = 0.003$). However, no linear relationships were evident during 1996 or 1998 (table 5 and fig. 1).

CONCLUSIONS

Several species, including species of conservation concern such as the black and white warbler, occurred throughout a variety of opening sizes. Additionally, most species had their highest densities recorded in group openings. Apparently these openings embedded within a more mature habitat matrix are attractive to a variety of species that utilize these early successional habitats. These openings also increase edge and habitat patchiness within a forested stand. However, the length of time a group opening maintains sufficient dissimilarity and edge contrast with the surrounding matrix habitat to sustain these relationships is unknown.

Total bird, migrant, and resident densities were only related with opening size during one of the three years of this study. This was probably due to the high numbers of birds recorded in the four clearcuts in 1995. No relationships between densities of individual species and opening size were apparent. The group-opening sizes in this study ranged from 0.54-2 to 62 ac, with the extremes differing by only 2.08 ac. This distribution of sizes may not be sufficient to reveal potential relationships between opening size and densities of individual species. Additionally, other factors such as opening shape and surrounding habitat matrix characteristics may also influence these relationships (Sisk and others 1997). We suggest additional research on a broader range of opening sizes.

Group-selection stands in our study are in a transition from an even-aged to an uneven-aged structure. Thus, bird community characteristics are expected to change over time as the vegetation composition and structure of these group openings and clearcuts change. Relative bird abundance was significantly higher within these same group-selection stands than for untreated controls in 1994 and 1998 (Thill and others, in press). Thus, our results suggest that during the early part of this transition, relative bird abundance may be enhanced by group-selection cuts in late-rotation, shortleaf pine dominated stands. However, before we can endorse this treatment, data are needed on reproductive success of birds utilizing these openings under a variety of successional stages.

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THE RELATIONSHIP BETWEEN BASAL AREA AND HARD MAST PRODUCTION IN THE OUACHITA MOUNTAINS

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Abstract—Because the relationship between stand density and hard mast production is not clear, we investigated the effects of varying total overstory basal area (BA) on acorn and hickory nut production in the Ouachita Mountains. We used Whitehead visual surveys to estimate mast production in oaks (*Quercus* spp.) and hickories (*Carya* spp.) located in 20 stands under five silvicultural treatments, each varying in residual BA. In 5 years of data (1994-1998), we found no linear relationship between BA and mast production for the red oak subgenus. A significant linear relationship existed between BA and hickory nut production two of the 5 years. A significant linear relationship existed between acorn production and BA for white (*Q. alba*) and post (*Q. stellata*) oaks all 5 years; trees in stands with lower BA had higher production indices. Because white and post oaks tend to be the dominant mast producers in pine-hardwood stands on south-facing slopes in the Ouachita Mountains, thinning these areas should generally increase mast production by residual trees. However, we did not measure the effects of thinning on stand-wide mast production.

INTRODUCTION

Hard mast production is an important element of forest ecology. Mast abundance affects both forest regeneration and wildlife that rely on mast as a food source. Many wildlife species are so dependent on hard mast that its supply may influence their condition, reproduction, movements, survival, and population parameters (McShea and Schwede 1993, Nixon and others 1975, Wentworth and others 1990).

For decades, researchers have investigated the factors influencing hard-mast (primarily acorns) production. Their efforts have provided considerable, and frequently conflicting, information on the variation in production among years, species, and individual trees. Moreover, these studies have provided many theories, from genetics to weather, to account for this variation (e.g., Christisen 1955, Christisen and Korschgen 1955, Downs and McQuilken 1944, Farmer 1981, Koenig and others 1996, Sork and others 1993). However, little is known on how forest management practices affect mast production.

Land managers who thin forest stands cite increased mast production as one of the benefits of thinning. Although the relationship between stand density and seed production has been thoroughly investigated in pines (*Pinus* spp.; e.g., Bilan 1960, Croker 1952, Godman 1962, Wenger 1954), information on oaks (*Quercus* spp.) and hickories (*Carya* spp.) is limited mostly to anecdotal and observational references (e.g., Gysel 1956, Minckler and McDermott 1960, Reid and Goodrum 1957, Sharp and Sprague 1967). Studies suggest a relationship may exist in oaks, but this relationship is poorly understood. For this study, we investigated the relationship between total (pine and hardwood) basal area (BA) and mast production by oaks and hickories to determine if reducing stand density increases the mast production of residual trees.

METHODS

Study Areas

Research was conducted in the 20 phase II wildlife stands of the USDA Forest Service Ouachita Mountain Ecosystem Management Research Project located in the Ouachita and Ozark National Forests of Arkansas and Oklahoma (Baker 1994). Sampling began the first year after four different silvicultural systems were applied to initially even-aged, pine-hardwood stands (approximately 40 acres each). Trees were surveyed in four replications of clearcut (with scattered overstory hardwoods retained for wildlife), single-tree selection, group selection, and shelterwood. Harvesting occurred in 1993. In addition, trees were surveyed in four closed-canopy, late-rotation unharvested stands (60-80 years old). For a complete description of stand treatments, see Baker (1994). Basal area estimates for trees ≥3.6 inches d.b.h. were derived using the method of Guldin and others (1994). Immediately after harvest, average total overstory BA within harvested portions of stands, excluding greenbelts, was 6.5 (1.3 pine and 5.2 hardwood) square feet per acre in clearcuts, 49.6 (36.1 pine and 13.5 hardwood) square feet per acre in shelterwoods, 67.5 (54.0 pine and 13.5 hardwood) square feet per acre in single-tree selection stands, 18.3 (1.3 pine and 17.0 hardwood) square feet per acre in group openings, and 88.8 (61.8 pine and 27.0 hardwood) square feet per acre in group selection matrixes. Average basal area among unharvested stands was 127.5 (98.8 pine and 28.7 hardwood) square feet per acre (Unpublished data. James M. Guldin, Forest Ecologist, USDA Forest Service, Southern Research Station, P.O. Box 1270, Hot Springs, AR 71902).

Production Estimates

In each stand, we sampled all oaks ≥7.9 in. d.b.h. and all hickories ≥5.9 in. d.b.h. (regardless of crown placement) located within 4 to 9 belt transects (depending on stand size and shape) that were parallel and 49.2-feet wide (Thill and

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Table 1—Mean (± SE) number of trees sampled in each stand or stand subtreatment, by species (or species group) and year, and the number of sample areas (N) included in the regression analysis between Whitehead mast indices and BA in the Ouachita Mountains, 1994–1998

	White oak		Post oak		Hickories		Red oak subgenus	
Year	Mean	N	Mean	N	Mean	N	Mean	N
1994	16.5 ± 1.86	13	17.4 ± 3.54	7	13.3 ± 1.37	8	20.5 ± 6.03	4
1995	18.8 ± 1.89	14	16.0 ± 3.26	10	15.5 ± 2.02	10	25.4 ± 8.73	5
1996	19.2 ± 2.32	14	17.4 ± 3.26	8	14.2 ± 1.42	10	20.0 ± 6.27	7
1997	22.1 ± 2.31	14	16.5 ± 3.08	10	17.8 ± 2.23	11	21.7 ± 7.98	6
1998	22.8 ± 2.40	14	16.8 ± 3.21	10	17.9 ± 2.26	11	19.9 ± 6.87	7

BA = basal area.

others 1994). Transects were 98 to 312 feet apart, perpendicular to slope contours, and >164 feet from the stand edge. Because residual mast-producing tree densities differed among treatment types, we sampled less area in stands with higher tree densities (unharvested, group selection matrix areas, and single-tree selection stands). Thus, total area sampled in each stand was 2.5-3.2 acres in unharvested stands, 2.3-3.5 acres in single-tree selection stands, 2.8-2.9 acres in group selection stands, 4.5-5.3 acres in shelterwood stands, and 5.1-5.3 acres in clearcut stands.

We sampled each stand in mid- to late-August of 1994-1998. Because of costs and biases associated with seed traps (Perry and Thill 1999), we derived indices of oak and hickory production using the Whitehead visual survey method (Whitehead 1969). These indices were derived by visually estimating percent of a tree's crown producing nuts, percent of twigs with nuts, and the average number of nuts per twig using binoculars. This method results in an index ranging from 0 (no production) to 10 (bumper crop) and can be useful for distinguishing differences in mast production among areas (Perry and Thill 1999).

Data Analysis

Within group selection stands, we considered group openings and the thinned matrix surrounding openings as separate treatments because of substantial differences in residual BA. Therefore, total number of stands and subtreatments was 24. We did not include trees located in greenbelts (unharvested buffers surrounding streams and drains) in the analysis; likewise, we excluded greenbelt BA from total stand BA calculations. We calculated a mean production index for each stand or subtreatment where at least eight trees of a species or species group existed within the sample area. We regressed each mean production index with the mean BA for that stand or subtreatment using linear regression (SAS Institute Inc. 1988) at the ≤0.10 level of probability. We regressed each year separately.

Oak and hickory species composition differed among stands because of site and/or geographic locality differences. Thus, post oaks (*Q. stellata* Wang.) were the dominant oak species in some stands but were absent in others, whereas white oaks (*Q. alba* L.) were the dominant oak in most stands. Furthermore, the densities of some mast species were too low to collect adequate sample sizes (≥8 trees per stand or subtreatment). Therefore, total number of areas used in regression equations did not reflect the total number of stands and subtreatments. Oaks of the red oak group (subgenus *Erythrobalanus*) were rare in most stands; therefore, we combined black (*Q. velutina* Lam.), northern red (*Q. rubra* L.), southern red (*Q. falcata* Michx.), and blackjack (*Q. marilandica* Muenchh.) into a single red oak group. We also combined mockernut hickory (*C. tomentosa* Nutt.) and black hickory (*C. texana* Buckl.) into a single hickory group. For each species or species group, the mean number of trees sampled ranged from 13.3-25.4 per stand or subtreatment and total number of stands or subtreatments in the analysis was 4-14 (table 1).

RESULTS AND DISCUSSION

Red Oak Subgenus

Among the combined species of the red-oak subgenus, we found no relationship between BA and production indices all 5 years (table 2). Healy (1997) found individual northern red oaks in New England stands thinned to 50 percent stocking produced more acorns than trees in unthinned stands. Paugh (1970) found individual red oaks in heavily thinned stands produced more mast than trees in unthinned stands, but trees in lightly thinned stands produced less mast than trees in unthinned areas. Harlow and Eikum (1963) found turkey oaks (*Q. laevis* Walt.) in stands thinned to 50 or 90 percent of their original BA produced more mast than trees in unthinned or 75 percent thinned stands. Although these studies suggest that heavy thinning promotes increased

Table 2—Statistics for the yearly relationship between BA and production indices for red oak species sampled in the Ouachita Mountains, 1994–1998

Year	*F*	*P*	df
1994	0.01	0.93	1,2
1995	0.02	0.89	1,3
1996	1.01	0.36	1,5
1997	0.00	0.99	1,4
1998	0.94	0.38	1,5

BA = basal area.

mast production, they also suggest a linear relationship between density and mast production may not exist for red oaks. However, grouping data from four species and small sample sizes may have adversely affected our analysis. Furthermore, much larger sample sizes are probably required to detect a relationship given the highly variable nature of mast production among individual trees, years, and areas.

White and Post Oaks

A significant relationship existed between BA and production indices in white oaks all 5 years of the study (fig. 1). Production indices decreased with increases in total overstory BA. This relationship was relatively weak ($r^2 = 0.31$) in 1994 and relatively strong ($r^2 = 0.71$) in 1995. A similar significant relationship existed between BA and production indices in post oaks all 5 years of the study (fig. 2), with the weakest relationship ($r^2 = 0.29$) occurring in 1997 and the strongest relationship ($r^2 = 0.63$) in 1996.

Yearly variations in acorn production did not appear to affect the strength of the relationship between production and BA. Among white oaks, the strongest relationship ($r^2 = 0.71$) was during a bumper-crop year (1995), whereas the weakest relationship was during an average production year (1994). The r^2 values in 1998 (a near mast-failure year), 1996 (an average year), and 1997 (an above average year) were similar. Yearly variation in production did not appear to affect the strength of the relationship in post oaks.

Equations describing the relationship between stand BA and mast production indices in white and post oaks differed each year, depending on the yearly level of production. Because of these yearly differences, we can only roughly calculate a predictive equation for the effect of BA on Whitehead index. An averaged equation for white oaks (all 5 years of data pooled) was: $y = 3.5-0.022x$, where y = the predicted mast index and x = BA (square feet per acre). For post oaks, the averaged equation was: $y = 4.3-0.030x$. For example, a total overstory BA of 25 square feet per acre will, on average, result in a white-oak index of 2.95, whereas a BA of 120 will average 0.86.

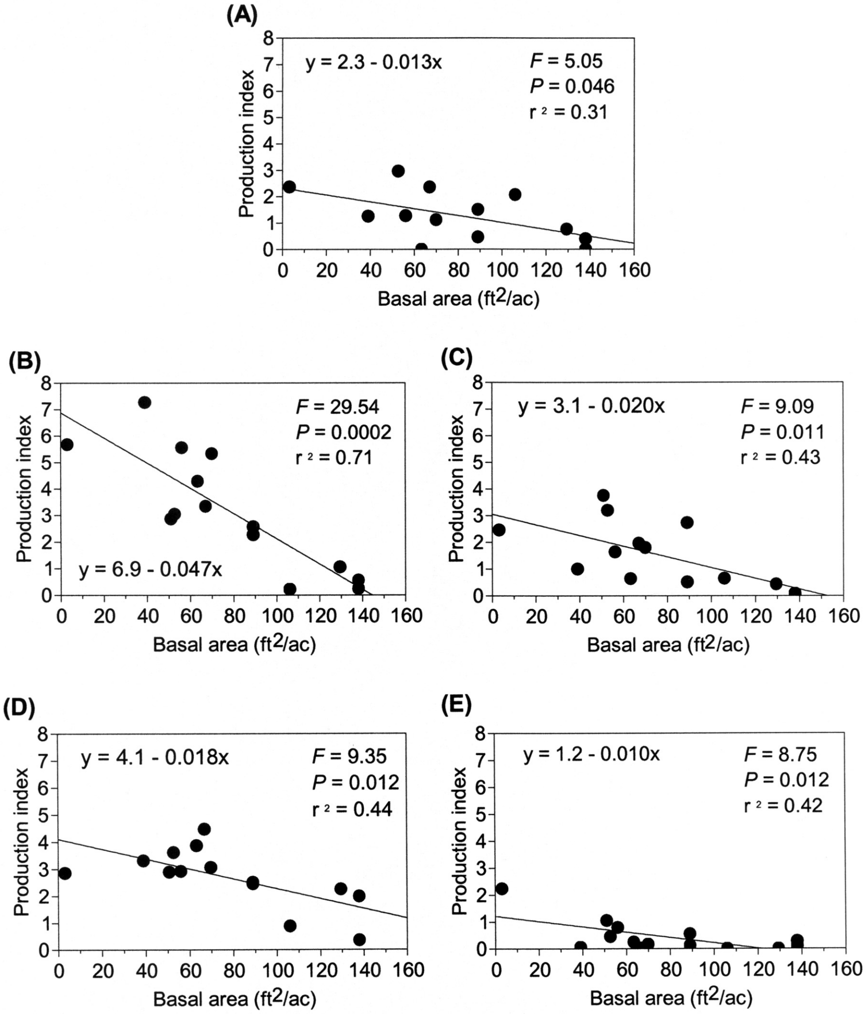

Figure 1—Yearly relationship (A = 1994, B = 1995, C = 1996, D = 1997 and E = 1998) between basal area and mean Whitehead mast production indices for white oaks (*Quercus alba*) in the Ouachita Mountains, 1994-1998.

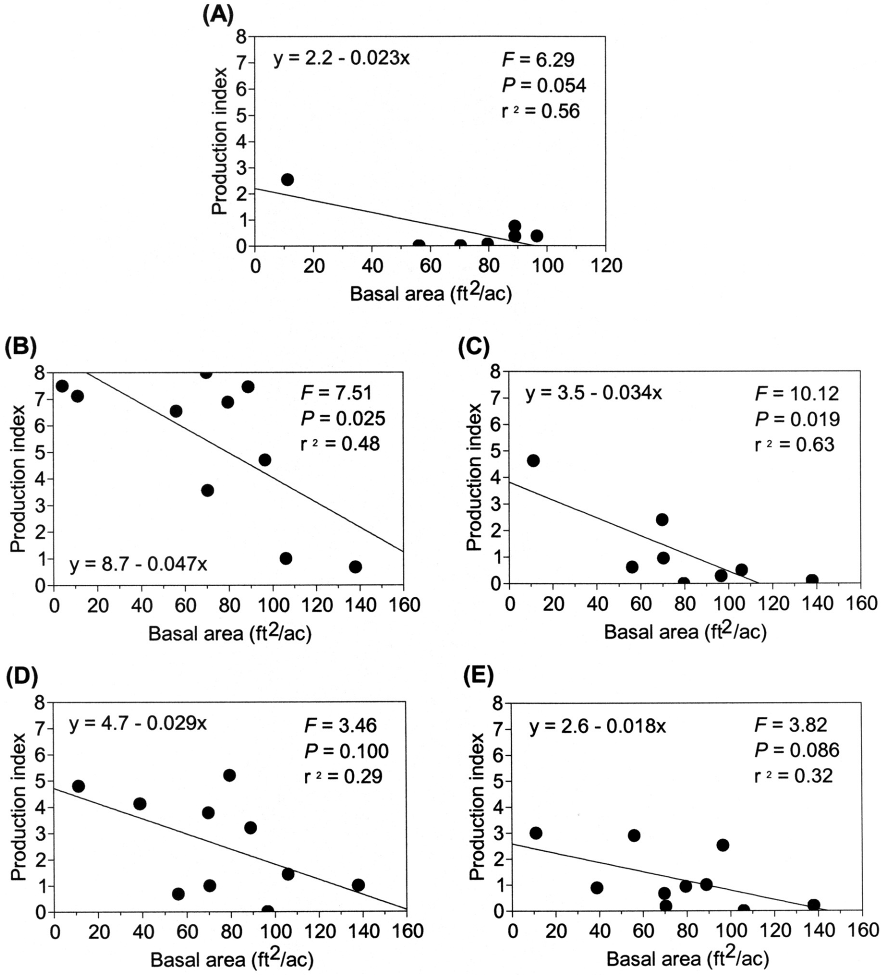

Figure 2—Yearly relationship (A = 1994, B = 1995, C = 1996, D = 1997 and E = 1998) between basal area and mean Whitehead mast production indices for post oaks (*Quercus stellata*) in the Ouachita Mountains, 1994-1998.

Hickories

A significant relationship existed between BA and hickory production indices only in 1995 and 1997 (fig. 3). No significant relationship occurred in 1994 ($F = 1.47$, $P = 0.27$, df = 1,6), 1996 ($F = 1.27$, $P = 0.29$, df = 1, 8), or 1998 ($F = 0.09$, $P = 0.78$, df = 1, 9). The relationship was strong ($r^2 = 0.79$) in 1995 (an above-average hickory-mast year) and moderate ($r^2 = 0.42$) in 1997 (also an above-average hickory-mast year). Because this relationship existed only two of the 5 sample years, the effect of stand density on hickory production is inconclusive.

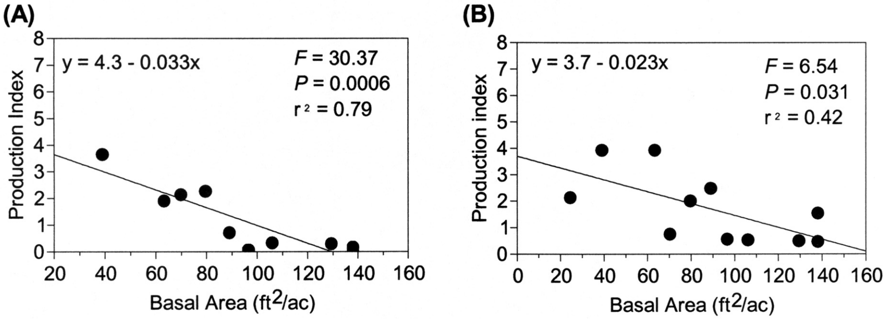

Figure 3—Yearly relationship (A = 1995, B = 1997) between basal area and mean Whitehead mast production indices for hickories (*Carya texana* and *C. tomentosa*) in 1995 and 1997 in the Ouachita Mountains. No significant linear relationship existed in 1994, 1996, or 1998 ($P > 0.10$).

CONCLUSIONS

Land managers strive to create optimal wildlife habitat by providing cover, breeding areas, and adequate resources such as food. Because hard mast is such an important food source for numerous species of wildlife, managers should consider optimizing mast production in conjunction with other management goals. Our study suggests thinning, even light thinning, can increase mast production in white and post oaks. Because these are two of the most abundant mast-producing species in pine-hardwood stands on south-facing slopes in the Ouachita Mountains, thinning should generally increase mast production of individual trees in these areas. However, we evaluated the effects only of stand density on individual trees within those stands, not the effects of thinning on overall stand production. Because thinning removes some mast-producing trees, the overall effects on stand-level production are unknown. Our results suggest production by hickories or red oaks may not benefit from thinning, but further investigation is needed to confirm this.

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INITIAL RESPONSE OF INDIVIDUAL SOFT MAST-PRODUCING PLANTS TO DIFFERENT FOREST REGENERATION METHODS IN THE OUACHITA MOUNTAINS

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Abstract—Recent policy changes have eliminated clearcutting as the primary pine regeneration method on Federal lands in the Southern United States. However, the effects of alternative natural regeneration methods on soft mast production are unknown. We compared plant coverage and mast production of 37 soft mast-producing plants among four regeneration methods (clearcut, shelterwood, single-tree selection, and group selection), and in mature, unharvested stands the first, third, and fifth years after timber harvest in the Ouachita Mountains of Oklahoma and Arkansas. Species richness and diversity of plants that produced mast were greatest in unharvested stands the first postharvest year, although total mast production did not differ among treatments. By the fifth postharvest year, total mast production and species richness were greater in harvested stands than in unharvested stands. Overall, shelterwoods most closely approximated the abundant soft mast production usually associated with clearcuts. Most of the 37 taxa investigated produced little or no mast, regardless of treatment. Mast from only seven taxa made up 48-100 percent of total production in each treatment. Pokeberry (*Phytolacca americana* L.) was the most abundant soft mast produced in intensively harvested areas (clearcuts, shelterwoods, and group openings) the first year, but production declined dramatically in subsequent years. Blackberry (*Rubus* spp.) was the most abundant mast in these areas by the fifth year. Muscadine grape (*Vitis rotundifolia* Michx.) was generally the most abundant mast in other treatments.

INTRODUCTION

Quality forest habitats provide not only the vegetation structure needed by wildlife, but also adequate resources such as food and water. Soft mast is an important component of forest wildlife habitat, comprising a great percentage of the seasonal diets of many wildlife species (Martin and others 1951). Soft mast from different plant species varies in nutritional quality (Halls 1977), and individual wildlife species may prefer certain types of mast. Thus, wildlife managers need to know what factors affect the abundance and production of individual soft mast species.

Recent policy changes by Federal land management agencies, such as the USDA Forest Service, have increased the use of forest regeneration methods other than clearcut and plant. Prior research provides substantial information on soft mast responses to clearcutting (e.g., Campo and Hurst 1980, Johnson and Landers 1978, Stransky and Halls 1980). However, the effects of alternative natural regeneration methods on individual soft mast-producing species are unknown. Therefore, we compared the initial production and coverage of 37 soft mast-producing species among four regeneration methods (clearcut, shelterwood, single-tree selection, and group selection) and in mature, unharvested forest stands (controls). We sampled stands the first (1994), third (1996), and fifth (1998) years after initial harvest in 1993. We present data for all 37 species but focus on the seven species that produced the most mast.

METHODS

Study Areas

We conducted the study in the 20 wildlife research stands of the USDA Forest Service Ecosystem Management (phase II) research study, located in the Ouachita and Ozark National Forests of Oklahoma and Arkansas (Baker 1994, Thill and others 1994). Five late-rotation, mixed pine-hardwood stands were selected in four physiographic zones (north, south, east, and west blocks) of the Ouachita Mountains (Baker 1994). Within each of the four blocks, stands randomly received one of five treatments. Treatments were single-tree selection, group selection, shelterwood, clearcut, and late-rotation unharvested. Timber harvesting was conducted in spring and summer of 1993; site preparation in natural regeneration stands occurred the following winter. Mechanical ripping of clearcuts (prior to pine planting) occurred the following summer (1994). For a detailed description of treatments see Baker (1994) and Perry and others (1999).

Unharvested buffer strips or greenbelts (typically 15 m on both sides of ephemeral and intermittent streams) were established for water quality protection within harvested stands. We considered greenbelts a subtreatment of harvested stands; we averaged greenbelt data from all 16 harvested stands for comparison with other treatments.

Soft Mast Sampling

Prior to timber harvest, in each stand we established 100 permanent sampling stations at 15-m intervals along 4-9 (depending on stand size and shape) parallel transects (Thill and others 1994). Transects were 30-95 m apart, oriented perpendicular to stand slope, and >50 m from the stand edge. We randomly selected a subsample of these 100 stations to estimate plant cover and soft mast production.

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Table 1—Soft-mast-producing taxa surveyed for production and coverage in 20 forest stands under various treatments in the Ouachita Mountains of Arkansas and Oklahoma during summer 1994, 1996, and 1998

Downy serviceberry (*Amelanchier arborea* Michx.f)[a]
Rattan (*Berchemia scandens* Hill)
Bumelia (*Bumelia lanuginosa* Michx.)[a]
American beautyberry (*Callicarpa americana* L.)
Hackberries (*Celtis* spp.)
Fringe tree (*Chionanthus virginicus L.)*
Redberry moonseed (*Cocculus carolinus* L.)
Flowering dogwood (*Cornus florida* L.)
Narrow-leafed dogwood (*Cornus obliqua* Raf.)
Hawthorns (*Crataegus* spp.)
Persimmon (*Diospyros virginiana* L.)
Deciduous holly (*Ilex decidua* Ait.)
American holly (*Ilex opaca* Ait.)
Yaupon holly (*Ilex vomitoria* Ait)[a]
Eastern redcedar (*Juniperus virginiana* L.)[a]
Partridgeberry (*Mitchella repens* L.)
Mulberry (*Morus rubra* L.)
Blackgum (*Nyssa sylvatica* Marsh.)[a]
Virginia creeper (*Parthenocissus quinquefolia* L.)[a]
Pokeberry (*Phytolacca americana* L.)[b]
Black cherry (*Prunus serotina* Ehrh.)
Wild plums (*Prunus* spp.)
Carolina buckthorn (*Rhamnus caroliniana* Walt.)
Fragrant sumac (*Rhus aromatica* Ait.)
Winged sumac (*Rhus copallina* L.)
Smooth sumac (*Rhus glabra* L.)
Wild roses (*Rosa* spp.)
Blackberries (*Rubus* spp.)
Sassafras (*Sassafras albidum* Nutt.)
Greenbriers (*Smilax* spp.)
Coralberry (*Symphoricarpos orbiculatus* Moench)
Poison ivy (*Toxicodendron radicans L.)*
Sparkleberry (*Vaccinium arboreum* Marsh.)
Blueberries (*Vaccinium* spp.)
Rusty blackhaw (*Viburnum rufidulum* Raf.)
Muscadine grape (*Vitis rotundifolia* Michx.)
Other grapes (*Vitis* spp.)

[a] No cover estimates were derived for these species in 1994.
[b] Herbaceous species

We estimated soft mast production and percent coverage, by species, during the summers of 1994, 1996, and 1998 (table 1). In 1996 and 1998, we included additional percent cover estimates for some species not measured in 1994, although we detected no production by these species throughout the study. In 1994, we sampled three 1-m^2 plots, located at 30 of the 100 sampling stations (90 m^2 sample area), in each unharvested and group selection stand. In each clearcut, shelterwood, and single-tree selection stand, we sampled one 1-m^2 plot at 40 of the stations (40 m^2 sample area). However, because we observed heterogenous distribution of soft mast within all stands in 1994, we increased sampling effort at all stations to one 3x3-m plot, located at each of 60 stations (540 m^2 total sample area in each stand) in 1996 and 1998. For further information on sampling effort by treatment and year, see Perry and others (1999).

We conducted soft mast surveys in mid-June, mid-July, and mid-August. During each sampling period, we measured a different set of species to coincide with ripening phenology of the major fruit-producing species. During sampling, we counted all soft mast, including green fruits, located within plots to a height of 2 m. To reduce potential bias from herbivory, we tallied evidence of removed fruits when possible. We developed wet to dry mass conversion factors by collecting, drying to constant mass, and weighing samples of each fruit type. We visually estimated percent coverage of each fruit-producing species at each plot in mid-July. A single observer estimated coverage in west- and south-block stands, and another observer estimated coverage in east- and north-block stands. Although we measured coverage of downy serviceberry (*Amelanchier arborea* Michx.f.), sampling occurred too late in the season to include soft mast from this early-spring producer.

For species with large seed heads containing numerous individual fruits [e.g., winged sumac (*Rhus copallina* L.)], we developed regression equations to convert volume estimates to mass. First, we collected 20-50 fruit heads per species. We then measured these fruit heads in three dimensions to determine volume, then measured mass to derive volume-to-mass estimates. During fruit surveys, we measured volume of each seed head on each plot to estimate mass produced.

Data Analysis

We derived means of soft mast production (kg/ha dry mass) and percent coverage for each soft mast species in each stand. We calculated treatment means from the four stands/treatment, except for clearcuts in 1994. In 1994, one clearcut was being ripped during the July surveys; therefore, we only included three clearcuts in the 1994 analysis. We used the same stands for sampling each year except for one unharvested stand. In 1997, the east-block unharvested stand was inadvertently harvested and subsequently replaced in 1998 with a similar stand. Because data for individual soft mast species among treatments were not parametric, we compared means among treatments using analysis of variance (ANOVA) on ranks and Duncan's multiple range test at the 0.10 level (SAS Institute Inc. 1988). Measured values, rather than ranks, are presented in all tables. However, because of ranking, some values are statistically lower than others even though the presented means are greater.

Group selection stands were comprised of group openings, where basal areas (BA) averaged 4.1 ± 0.7 m^2/ha, and the surrounding forest matrix, which was thinned to 20.3 ± 0.8 m^2/ha. Because of notable differences in BA and mast production between the openings and surrounding matrices,

Table 2—Mean (± SE) percent cover of soft-mast-producing plants, by treatment and habitat types within treatments, measured during July 1994 (the first post-harvest year) in the Ouachita Mountains

				Group selection[c]				
Species	Unharvested	Greenbelt[b]	Single tree selection	Matrix	Openings	Combined	Shelterwood	Clearcut
American beautyberry	0.03 ± 0.03	0.06 ± 0.06	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.13 ± 0.13	0.00 ± 0.00
American holly	0.15 ± 0.15	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Blackberries	0.04b[a] ± 0.04	0.17b ± 0.17	0.13ab ± 0.08	0.22 ± 0.11	0.31 ± 0.24	0.25a ± 0.12	1.35a ± 0.59	0.09ab ± 0.06
Black cherry	0.10 ± 0.08	1.59 ± 1.32	0.19 ± 0.15	0.07 ± 0.07	0.02 ± 0.01	0.04 ± 0.04	0.00 ± 0.00	0.34 ± 0.27
Blueberries	10.06 ± 3.01	2.19 ± 1.36	5.66 ± 3.22	1.49 ± 1.13	1.22 ± 0.43	1.38 ± 0.87	1.54 ± 1.07	2.10 ± 1.79
Flowering dogwood	10.96 ± 5.44	4.01 ± 1.46	6.47 ± 3.51	2.03 ± 0.44	1.01 ± 0.59	1.90 ± 0.35	3.12 ± 1.47	0.68 ± 0.60
Fragrant sumac	0.00 ± 0.00	0.00 ± 0.00	0.06 ± 0.06	0.00 ± 0.00	0.01 ± 0.01	<0.01	0.00 ± 0.00	0.00 ± 0.00
Greenbriers	2.86 ± 1.51	2.05 ± 1.06	0.49 ± 0.14	0.81 ± 0.23	0.75 ± 0.75	0.95 ± 0.35	0.95 ± 0.47	0.22 ± 0.20
Hackberries	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.06 ± 0.06	0.00 ± 0.00	0.06 ± 0.06	0.00 ± 0.00	0.00 ± 0.00
Hawthorns	0.07b ± 0.07	0.02b ± 0.02	0.22ab ± 0.21	0.21 ± 0.13	0.00 ± 0.00	0.31a ± 0.15	0.00b ± 0.00	0.00b ± 0.00
Muscadine grape	2.64 ± 0.71	2.28 ± 0.80	1.76 ± 0.84	5.11 ± 3.60	6.93 ± 6.32	5.05 ± 3.74	4.25 ± 1.83	1.37 ± 1.35
Other grapes	0.43 ± 0.28	0.02 ± 0.02	0.01 ± 0.01	0.72 ± 0.62	0.24 ± 0.24	0.54 ± 0.47	0.35 ± 0.35	0.10 ± 0.10
Partridgeberry	<0.01	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Persimmon	0.08 ± 0.08	0.38 ± 0.38	0.01 ± 0.01	0.00 ± 0.00	0.02 ± 0.02	<0.01	0.00 ± 0.00	0.00 ± 0.00
Poison ivy	2.86 ± 0.60	5.19 ± 1.43	4.67 ± 2.41	3.69 ± 1.70	3.94 ± 1.76	3.73 ± 1.42	1.87 ± 0.19	1.97 ± 1.10
Pokeberry	0.00b ± 0.00	0.08b ± 0.08	0.63b ± 0.63	0.02 ± 0.02	1.43 ± 1.43	0.37b ± 0.37	1.46a ± 1.25	1.63a ± 1.37
Rattan	0.02 ± 0.02	0.41 ± 0.22	0.08 ± 0.08	0.18 ± 0.17	0.08 ± 0.08	0.17 ± 0.12	0.00 ± 0.00	0.00 ± 0.00
Rusty blackhaw	0.00 ± 0.00	1.03 ± 0.76	0.07 ± 0.06	0.02 ± 0.02	0.00 ± 0.00	0.01 ± 0.01	0.00 ± 0.00	0.11 ± 0.10
Sassafras	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.02 ± 0.02	0.00 ± 0.00	0.01 ± 0.01	0.00 ± 0.00	0.00 ± 0.00
Smooth sumac	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.05 ± 0.05	0.00 ± 0.00
Sparkleberry	1.85 ± 1.16	1.81 ± 1.35	0.29 ± 0.25	2.08 ± 1.56	0.42 ± 0.42	1.86 ± 1.46	0.00 ± 0.00	0.01 ± 0.01
Wild plums	<0.01b	0.07b ± 0.07	0.11ab ± 0.07	0.12 ± 0.11	0.00 ± 0.00	0.06ab ± 0.05	0.00b ± 0.00	0.92a ± 0.77
Winged sumac	0.00b ± 0.00	0.10b ± 0.08	0.01b ± 0.01	0.11 ± 0.09	0.71 ± 0.68	0.32a ± 0.21	0.47a ± 0.16	0.59a ± 0.30

[a] Within rows, means followed by the same letter were not different ($P < 0.10$) using ANOVA on ranks and Duncan's MRT.

[b] Averaged across all harvested stands which contained greenbelt plots ($N = 15$).

[c] Only combined values were included in the ANOVA.

we present data for these two habitats separately. However, only combined data (derived from stand-wide averages) were included in the ANOVA analysis. We also determined total soft mast production and coverage in the greenbelts of each stand (*n* = 15 stands in 1994 and *n* = 16 stands in 1996 and 1998). We then determined mean soft mast production and coverage in greenbelts by averaging across all stands.

We calculated richness, evenness, and diversity of species which produced mast and compared these measures, by year, among the five primary treatments (clearcut, shelterwood, single-tree selection, group selection, and unharvested) using analysis of variance and Duncan's multiple range test at the 0.10 level (SAS Institute Inc. 1988). Richness was defined as the total number of species producing soft mast per stand. Species diversity was the Shannon index (Shannon and Weiner 1949); Pielou's J was our evenness measure (Pielou 1969).

RESULTS AND DISCUSSION

Soft mast plant coverage was relatively low in harvested stands the first postharvest year, apparently due to logging disturbance (table 2). Only nine species produced mast among all stands in 1994 (table 3). In group openings and clearcut stands, pokeberry (*Phytolacca americana* L.) was the only mast produced. In the third postharvest year (1996), coverage of most plant species increased in harvested stands (table 4). Overall soft mast production in the third year was greater in harvested stands than in unharvested stands (Perry and others 1999). In the third year, the number of species producing mast increased to 16 (table 5). By the fifth postharvest year, coverage of many plant species was greater in harvested stands than in unharvested stands or greenbelts (table 6). Twenty-five species produced mast the fifth year, although many of these species produced only trace amounts in a single stand (table 7). Total production in the fifth year was greatest in clearcut and shelterwood stands, intermediate in single-tree selection and group selection stands, and lowest in unharvested stands; shelterwood and clearcut stands produced more than twice as much mast as single-tree selection and group selection stands (Perry and others 1999).

Species richness was greatest in unharvested stands the first postharvest year (table 8). In the third year, no difference existed in richness among treatments. By the fifth year, richness was greater in harvested stands than in unharvested stands. Diversity of mast-producing species was greatest in unharvested stands, and evenness was greatest in unharvested and group selection stands the first year. However, differences were not significant ($P > 0.10$) in subsequent years.

Pokeberry, blackberries (*Rubus* spp.), blueberries (*Vaccinium* spp.), muscadine grape (*Vitis rotundifolia* Michx.), other grapes (*Vitis* spp.), winged sumac, and American beautyberry (*Callicarpa americana* L.) were the seven most abundant soft mast producers during the study (table 9). Therefore, the following discussion focuses on these seven taxa.

Table 3—Mean (± SE) dry-weight of soft mast (kg/ha) produced, by treatment and habitat types within treatments, measured during summer 1994 (the first post-harvest year) in the Ouachita Mountains

Species	Unharvested	Greenbelt[b]	Single tree selection	Group selection[c] Matrix	Group selection[c] Openings	Group selection[c] Combined	Shelterwood	Clearcut
Blueberries	0.58a[a] ± 0.52	0.00b ± 0.00	0.00b ± 0.00	0.09 ± 0.09	0.00 ± 0.00	0.07b ± 0.07	0.07b ± 0.07	0.00b ± 0.00
Flowering dogwood	1.42 ± 0.98	1.13 ± 1.00	0.23 ± 0.23	0.71 ± 0.42	0.00 ± 0.00	0.47 ± 0.30	0.00 ± 0.00	0.00 ± 0.00
Greenbriers	0.25a ± 0.18	0.02b ± 0.02	0.00b ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00b ± 0.00	0.00b ± 0.00	0.00b ± 0.00
Muscadine grape	0.52a ± 0.29	2.06b ± 2.06	0.22b ± 0.22	0.00 ± 0.00	0.00 ± 0.00	0.00b ± 0.00	3.33b ± 3.33	0.00b ± 0.00
Other grapes	0.46 ± 0.46	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Poison ivy	0.02 ± 0.01	0.04 ± 0.04	0.00 ± 0.00	0.02 ± 0.02	0.00 ± 0.00	0.01 ± 0.01	0.00 ± 0.00	0.00 ± 0.00
Pokeberry	0.00b ± 0.00	2.57b ± 2.57	0.00b ± 0.00	0.00 ± 0.00	13.13 ± 13.13	3.28b ± 3.28	26.51a ± 21.60	23.05a ± 18.56
Rattan	0.00 ± 0.00	0.04 ± 0.04	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Sparkleberry	0.00 ± 0.00	0.01 ± 0.01	0.00 ± 0.00	0.02 ± 0.02	0.00 ± 0.00	0.02 ± 0.02	0.00 ± 0.00	0.00 ± 0.00

[a] Within rows, means followed by the same letter were not different (P < 0.10) using ANOVA on ranks and Duncan's MRT.
[b] Averaged across all harvested stands which contained greenbelt plots (N = 15).
[c] Only combined values were included in the ANOVA.

Table 4—Mean (± SE) percent cover of soft-mast-producing plants, by treatment and habitat types within treatments, measured during July 1996 (the third post-harvest year) in the Ouachita Mountains

			Single tree	Group selection[c]				
Species	Unharvested	Greenbelt[b]	selection	Matrix	Openings	Combined	Shelterwood	Clearcut
American beautyberry	0.03ab[a] ± 0.02	0.00b ± 0.00	0.06ab ± 0.05	0.01 ± 0.01	0.42 ± 0.27	0.08ab ± 0.05	0.80a ± 0.78	0.13a ± 0.08
American holly	0.07 ± 0.07	0.06 ± 0.06	0.08 ± 0.08	0.00 ± 0.00	1.38 ± 1.38	0.31 ± 0.31	0.01 ± 0.01	0.00 ± 0.00
Blackberries	0.08b ± 0.02	0.48b ± 0.31	2.08a ± 0.64	0.48 ± 0.10	8.92 ± 2.01	2.59a ± 0.84	7.86a ± 4.02	4.07a ± 1.43
Black cherry	0.17b ± 0.12	0.69b ± 0.31	1.76a ± 0.87	0.19 ± 0.10	0.00 ± 0.00	0.12b ± 0.07	1.48ab ± 0.48	1.39ab ± 1.12
Blackgum	4.54a ± 1.88	1.04bc ± 0.44	2.61ab ± 1.27	0.32 ± 0.07	0.57 ± 0.48	0.39bc ± 0.15	2.41abc ± 1.49	0.69c ± 0.68
Blueberries	10.40 ± 2.70	3.66 ± 1.35	7.66 ± 5.16	2.88 ± 2.24	2.41 ± 0.88	2.63 ± 1.77	2.78 ± 1.76	2.46 ± 1.36
Bumelia	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.15 ± 0.15	0.05 ± 0.05	0.00 ± 0.00	0.00 ± 0.00
Carolina buckthorn	0.04 ± 0.04	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.05 ± 0.05	0.00 ± 0.00
Coralberry	0.00 ± 0.00	0.00 ± 0.00	0.03 ± 0.03	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Downy Serviceberry	0.19a ± 0.07	0.06b ± 0.06	0.07b ± 0.07	0.00 ± 0.00	0.00 ± 0.00	0.19b ± 0.19	0.00b ± 0.00	0.00b ± 0.00
Eastern redcedar	0.06 ± 0.06	0.10 ± 0.07	0.05 ± 0.05	0.27 ± 0.27	0.00 ± 0.00	0.14 ± 0.14	0.12 ± 0.12	0.00 ± 0.00
Flowering dogwood	4.52 ± 1.06	4.11 ± 0.80	8.52 ± 4.28	2.43 ± 0.83	2.04 ± 0.86	2.33 ± 0.73	3.33 ± 1.02	1.03 ± 0.91
Fragrant sumac	0.15ab ± 0.15	0.01ab ± 0.01	0.18a ± 0.11	0.16 ± 0.12	0.42 ± 0.27	0.20a ± 0.12	0.19a ± 0.13	0.00b ± 0.00
Fringe tree	0.00 ± 0.00	0.25 ± 0.25	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Greenbriers	4.61 ± 2.93	2.18 ± 0.47	1.75 ± 0.61	1.35 ± 0.48	1.87 ± 1.56	1.60 ± 0.82	2.57 ± 0.93	0.98 ± 0.53
Hackberries	0.00 ± 0.00	0.05 ± 0.05	0.04 ± 0.04	0.01 ± 0.01	0.00 ± 0.00	0.01 ± 0.01	0.00 ± 0.00	0.00 ± 0.00
Hawthorns	0.08 ± 0.07	0.19 ± 0.14	0.28 ± 0.10	0.21 ± 0.13	0.31 ± 0.24	0.30 ± 0.15	0.11 ± 0.09	0.05 ± 0.05
Mulberry	0.00 ± 0.00	0.00 ± 0.00	0.01 ± 0.01	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Muscadine grape	3.76b ± 1.25	3.24b ± 0.89	6.84ab ± 2.36	6.56 ± 3.46	13.22 ± 7.77	8.11b ± 4.37	17.88a ± 2.79	4.77b ± 2.98
Narrow-leaf dogwood	0.17 ± 0.17	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Other grapes	0.43ab ± 0.23	0.05c ± 0.02	0.74a ± 0.28	0.68 ± 0.50	1.17 ± 0.48	0.71ab ± 0.39	0.86ab ± 0.33	0.26bc ± 0.22
Partridgeberry	0.49a ± 0.30	0.00b ± 0.00	0.00b ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00b ± 0.00	0.00b ± 0.00	0.00b ± 0.00
Persimmon	0.06 ± 0.02	0.24 ± 0.11	0.21 ± 0.06	0.41 ± 0.19	2.20 ± 1.71	0.72 ± 0.36	0.34 ± 0.26	0.17 ± 0.15
Poison ivy	4.50 ± 1.47	9.15 ± 2.48	12.62 ± 7.10	10.11 ± 4.74	14.26 ± 5.53	10.69 ± 4.43	10.07 ± 1.76	7.68 ± 5.56
Pokeberry	0.00b ± 0.00	0.00b ± 0.00	0.15a ± 0.13	0.06 ± 0.06	1.22 ± 0.78	0.26a ± 0.14	0.37a ± 0.19	1.60a ± 1.28
Rattan	0.07 ± 0.05	0.14 ± 0.06	0.23 ± 0.18	0.01 ± 0.01	0.00 ± 0.00	<0.01	0.22 ± 0.19	0.00 ± 0.00
Redberry moonseed	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.11 ± 0.11	0.00 ± 0.00
Rusty blackhaw	0.02 ± 0.02	0.59 ± 0.37	0.14 ± 0.14	0.06 ± 0.04	0.03 ± 0.03	0.14 ± 0.08	0.01 ± 0.01	0.01 ± 0.01
Sassafras	0.02c ± 0.02	0.01c ± 0.01	0.11ab ± 0.07	0.06 ± 0.03	1.58 ± 1.37	0.45a ± 0.23	0.00c ± 0.00	0.15bc ± 0.15
Smooth sumac	0.00b ± 0.00	0.00b ± 0.00	0.00b ± 0.00	0.00 ± 0.00	0.14 ± 0.14	0.02ab ± 0.02	0.09a ± 0.06	0.10ab ± 0.10
Sparkleberry	0.73 ± 0.52	1.47 ± 0.85	1.11 ± 0.34	2.82 ± 2.17	3.38 ± 3.38	3.00 ± 2.51	0.81 ± 0.61	0.78 ± 0.22
Virginia creeper	2.82 ± 2.39	1.52 ± 0.48	2.07 ± 0.87	1.62 ± 0.50	2.94 ± 1.75	1.85 ± 0.57	4.17 ± 2.31	0.73 ± 0.61
Wild plums	0.03b ± 0.03	0.09b ± 0.07	0.33ab ± 0.22	0.64 ± 0.37	0.84 ± 0.56	0.79a ± 0.45	0.60 ± 0.43	0.64a ± 0.35
Wild rose	0.00c ± 0.00	0.05bc ± 0.03	0.03bc ± 0.03	0.38 ± 0.27	0.58 ± 0.55	0.37a ± 0.30	0.15ab ± 0.09	0.00c ± 0.00
Winged sumac	0.01c ± 0.01	0.38c ± 0.37	0.62b ± 0.27	0.26 ± 0.09	3.93 ± 2.39	1.38ab ± 0.81	3.10a ± 0.04	4.62a ± 2.46

[a] Within rows, means followed by the same letter were not different ($P < 0.10$) using ANOVA on ranks and Duncan's MRT.

[b] Averaged across all harvested stands which contained greenbelt plots ($N = 16$).

[c] Only combined values were included in the ANOVA.

Table 5—Mean (± SE) dry-weight of soft mast (kg/ha) produced, by treatment and habitat types within treatments, measured during summer 1996 (the third post-harvest year) in the Ouachita Mountains

Species	Unharvested	Greenbelt[b]	Single tree selection	Group selection[c]			Shelterwood	Clearcut
				Matrix	Openings	Combined		
American beautyberry	0.00 ± 0.00	0.00 ± 0.00	0.12 ± 0.12	0.00 ± 0.00	0.01 ± 0.01	<0.01	0.71 ± 0.71	0.03 ± 0.03
Blackberries	0.00d[a] ± 0.00	0.01d ± 0.01	0.20bc ± 0.05	0.05 ± 0.02	1.51 ± 1.34	0.51c ± 0.46	3.15a ± 2.08	1.17ab ± 0.80
Blueberries	0.04 ± 0.02	0.10 ± 0.06	0.22 ± 0.12	0.08 ± 0.08	<0.01	0.06 ± 0.06	0.76 ± 0.74	0.76 ± 0.74
Carolina buckthorn	<0.01	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	<0.01	0.00 ± 0.00
Flowering dogwood	0.02 ± 0.02	0.11 ± 0.09	0.17 ± 0.13	0.03 ± 0.03	0.10 ± 0.10	0.06 ± 0.03	0.10 ± 0.10	0.00 ± 0.00
Greenbriers	0.13 ± 0.10	0.02 ± 0.01	0.94 ± 0.94	0.02 ± 0.01	0.48 ± 0.48	0.18 ± 0.17	0.75 ± 0.44	0.01 ± 0.01
Muscadine grape	0.09b ± 0.04	1.00b ± 0.65	6.65a ± 2.03	0.53 ± 0.30	4.46 ± 3.94	1.76b ± 1.42	10.25a ± 5.60	0.95b ± 0.52
Other grapes	0.43ab ± 0.43	0.00b ± 0.00	0.20ab ± 0.20	0.00 ± 0.00	0.51 ± 0.51	0.09ab ± 0.09	1.59a ± 0.96	0.00b ± 0.00
Poison ivy	0.03bc ± 0.02	0.01c ± 0.01	0.13a ± 0.06	0.02 ± 0.01	0.00 ± 0.00	0.01bc ± 0.01	0.21ab ± 0.15	0.34bc ± 0.34
Pokeberry	0.00d ± 0.00	0.00d ± 0.00	0.64cd ± 0.64	0.02 ± 0.02	9.72 ± 6.55	2.11bc ± 1.41	1.92a ± 1.00	28.74ab ± 24.98
Rattan	0.00 ± 0.00	0.00 ± 0.00	<0.01	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Redberry moonseed	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.02 ± 0.02	0.00 ± 0.00
Sassafras	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	1.40 ± 1.40	0.24 ± 0.24	0.00 ± 0.00	0.00 ± 0.00
Sparkleberry	0.00 ± 0.00	0.00 ± 0.00	0.01 ± 0.01	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	<0.01
Wild rose	0.00b ± 0.00	<0.01b	0.00b ± 0.00	0.15 ± 0.13	0.23 ± 0.23	0.16a ± 0.15	0.29a ± 0.20	0.00b ± 0.00
Winged sumac	0.00c ± 0.00	0.32c ± 0.32	0.00c ± 0.00	0.52 ± 0.52	0.00 ± 0.00	0.40bc ± 0.40	4.25ab ± 4.06	8.86a ± 8.15

[a] Within rows, means followed by the same letter were not different ($P < 0.10$) using ANOVA on ranks and Duncan's MRT.

[b] Averaged across all harvested stands which contained greenbelt plots ($N = 16$).

[c] Only combined values were included in the ANOVA.

Table 6—Mean (± SE) percent cover of soft-mast-producing plants, by treatment and habitat types within treatments, measured during July 1998 (the fifth post-harvest year) in the Ouachita Mountains

			Single tree	Group selection[c]				
Species	Unharvested	Greenbelt[b]	selection	Matrix	Openings	Combined	Shelterwood	Clearcut
American beautyberry	0.05bc[a] ± 0.02	0.63c ± 0.63	0.04bc ± 0.02	0.07 ± 0.06	0.98 ± 0.47	0.24ab ± 0.09	1.93a ± 1.64	0.26bc ± 0.15
American holly	0.06 ± 0.05	0.65 ± 0.62	0.10 ± 0.09	0.19 ± 0.19	1.32 ± 1.23	0.40 ± 0.26	0.03 ± 0.03	0.00 ± 0.00
Blackberries	0.27c ± 0.13	1.39c ± 0.66	3.34b ± 0.84	2.04 ± 1.12	20.84 ± 4.18	6.76ab ± 2.45	14.08a ± 4.32	14.60a ± 1.51
Black cherry	0.33 ± 0.08	1.48 ± 0.65	2.05 ± 1.57	0.83 ± 0.46	3.43 ± 2.06	1.52 ± 0.88	3.29 ± 1.92	2.69 ± 1.34
Blackgum	4.42 ± 1.89	2.26 ± 0.71	4.17 ± 1.69	1.38 ± 0.72	6.88 ± 6.45	2.28 ± 1.65	3.26 ± 2.04	1.11 ± 0.73
Blueberries	10.25 ± 2.60	3.94 ± 1.39	7.51 ± 4.14	3.10 ± 2.07	3.08 ± 1.63	2.91 ± 1.80	3.52 ± 2.02	3.97 ± 1.71
Bumelia	0.00 ± 0.00	0.63 ± 0.62	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Carolina buckthorn	0.24 ± 0.24	0.84 ± 0.65	0.10 ± 0.10	0.04 ± 0.04	0.01 ± 0.01	0.03 ± 0.03	0.46 ± 0.46	0.00 ± 0.00
Coralberry	0.00 ± 0.00	0.63 ± 0.63	0.04 ± 0.04	0.00 ± 0.00	0.01 ± 0.01	<0.01	0.00 ± 0.00	0.00 ± 0.00
Deciduous holly	0.00 ± 0.00	0.89 ± 0.65	0.00 ± 0.00	0.00 ± 0.00	0.22 ± 0.22	0.07 ± 0.07	0.00 ± 0.00	0.00 ± 0.00
Downy Serviceberry	0.07 ± 0.04	0.64 ± 0.62	0.00 ± 0.00	0.00 ± 0.00	0.03 ± 0.03	0.01 ± 0.01	0.06 ± 0.06	0.00 ± 0.00
Eastern redcedar	0.13 ± 0.10	0.66 ± 0.32	0.15 ± 0.11	1.26 ± 0.76	0.50 ± 0.40	0.92 ± 0.43	0.56 ± 0.24	0.08 ± 0.07
Flowering dogwood	4.35 ± 1.38	5.56 ± 0.97	9.52 ± 4.85	3.81 ± 0.87	2.37 ± 1.00	3.49 ± 0.70	4.97 ± 1.15	1.75 ± 1.38
Fragrant sumac	0.13 ± 0.10	0.94 ± 0.68	0.33 ± 0.19	0.10 ± 0.07	1.56 ± 0.97	0.34 ± 0.21	0.56 ± 0.33	0.01 ± 0.01
Fringe tree	0.00 ± 0.00	1.00 ± 0.71	0.03 ± 0.03	0.01 ± 0.01	0.00 ± 0.00	0.01 ± 0.01	0.00 ± 0.00	0.01 ± 0.01
Greenbriers	4.42 ± 2.02	5.91 ± 1.47	2.21 ± 0.36	4.03 ± 2.06	4.77 ± 4.39	4.41 ± 2.74	5.75 ± 2.24	2.51 ± 1.73
Hackberries	0.00 ± 0.00	0.66 ± 0.62	0.07 ± 0.07	0.01 ± 0.01	0.03 ± 0.03	0.01 ± 0.01	0.00 ± 0.00	0.00 ± 0.00
Hawthorns	0.19 ± 0.06	0.75 ± 0.62	0.46 ± 0.13	0.27 ± 0.14	0.42 ± 0.26	0.34 ± 0.17	0.21 ± 0.06	0.29 ± 0.20
Mulberry	0.00 ± 0.00	0.66 ± 0.62	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.17 ± 0.10	0.03 ± 0.03
Muscadine grape	6.55 ± 2.34	6.87 ± 2.03	8.43 ± 3.15	11.24 ± 5.97	10.30 ± 7.13	10.77 ± 5.92	22.42 ± 4.20	7.52 ± 4.17
Narrow-leaf dogwood	0.09 ± 0.09	0.63 ± 0.63	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Other grapes	0.61ab ± 0.34	0.75b ± 0.62	0.95a ± 0.37	0.89 ± 0.47	2.32 ± 1.33	1.12ab ± 0.59	1.79a ± 0.62	0.70ab ± 0.53
Partridgeberry	0.49a ± 0.30	0.63b ± 0.63	0.00b ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00b ± 0.00	0.00b ± 0.00	0.00b ± 0.00
Persimmon	0.07 ± 0.04	0.73 ± 0.62	0.10 ± 0.06	0.36 ± 0.30	1.26 ± 0.72	0.58 ± 0.40	0.17 ± 0.17	0.04 ± 0.02
Poison ivy	6.82 ± 1.97	10.84 ± 2.39	16.61 ± 7.80	13.19 ± 5.94	20.62 ± 6.79	14.52 ± 5.85	15.89 ± 2.80	8.60 ± 5.17
Pokeberry	0.00b ± 0.00	0.63b ± 0.63	0.07b ± 0.07	0.00 ± 0.00	0.13 ± 0.13	0.03b ± 0.03	0.00b ± 0.00	0.22a ± 0.14
Rattan	0.20 ± 0.16	1.56 ± 0.77	0.41 ± 0.19	0.32 ± 0.18	1.04 ± 1.04	0.77 ± 0.51	0.61 ± 0.53	0.18 ± 0.18
Redberry moonseed	0.01 ± 0.01	0.65 ± 0.62	0.01 ± 0.01	0.01 ± 0.01	0.00 ± 0.00	0.01 ± 0.01	0.17 ± 0.16	0.00 ± 0.00
Rusty blackhaw	0.08 ± 0.05	1.26 ± 0.63	0.48 ± 0.36	0.10 ± 0.04	0.26 ± 0.15	0.15 ± 0.03	0.26 ± 0.15	0.34 ± 0.32
Sassafras	0.04 ± 0.02	0.68 ± 0.62	0.14 ± 0.07	0.19 ± 0.11	1.06 ± 0.67	0.46 ± 0.19	0.13 ± 0.09	0.35 ± 0.35
Smooth sumac	0.00c ± 0.00	0.63c ± 0.63	0.07bc ± 0.07	0.00 ± 0.00	0.42 ± 0.27	0.09abc ± 0.05	0.12ab ± 0.05	0.95a ± 0.84
Sparkleberry	1.03 ± 0.52	2.03 ± 0.89	1.38 ± 0.48	2.81 ± 2.07	4.24 ± 2.58	2.70 ± 1.52	1.26 ± 0.73	1.31 ± 0.55
Virginia creeper	4.50 ± 2.84	2.19 ± 0.74	2.54 ± 0.72	2.31 ± 0.92	4.13 ± 2.33	2.70 ± 1.13	4.98 ± 2.76	0.55 ± 0.41
Wild plums	0.27 ± 0.08	1.58 ± 0.73	1.55 ± 0.28	0.86 ± 0.49	1.08 ± 0.64	1.88 ± 1.23	1.26 ± 0.85	0.98 ± 0.81
Wild rose	0.05 ± 0.04	0.78 ± 0.62	0.08 ± 0.05	0.26 ± 0.19	1.11 ± 1.11	0.38 ± 0.33	0.40 ± 0.26	0.06 ± 0.06
Winged sumac	0.05b ± 0.05	1.21b ± 0.77	1.35a ± 0.39	0.47 ± 0.19	6.88 ± 3.49	2.47a ± 1.34	5.21a ± 0.68	9.04a ± 4.52
Yaupon holly	0.00 ± 0.00	0.63 ± 0.63	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00

[a] Within rows, means followed by the same letter were not different ($P < 0.10$) using ANOVA on ranks and Duncan's MRT.

[b] Averaged across all harvested stands which contained greenbelt plots (N = 16).

[c] Only combined values were included in the ANOVA.

Table 7—Mean (± SE) dry-weight of soft mast (kg/ha) produced, by treatment and habitat types within treatments, measured during summer 1998 (the fifth post-harvest year) in the Ouachita Mountains

			Single tree	Group selection[c]				
Species	Unharvested	Greenbelt[b]	selection	Matrix	Openings	Combined	Shelterwood	Clearcut
American beautyberry	<0.01bc[a]	0.00 ± 0.00c	0.14bc ± 0.14	0.00 ± 0.00	3.42 ± 3.17	0.57ab ± 0.53	3.61a ± 3.36	0.45ab ± 0.31
American holly	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	<0.01	0.00 ± 0.00
Blackberries	0.00c ± 0.00	0.04c ± 0.02	1.46b ± 0.66	0.90 ± 0.37	82.34 ± 23.45	18.31a ± 4.02	46.10a ± 32.37	48.37a ± 20.85
Black cherry	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	1.67 ± 1.67
Blueberries	0.50 ± 0.49	0.39 ± 0.20	0.11 ± 0.04	0.40 ± 0.40	0.78 ± 0.76	0.48 ± 0.47	2.63 ± 2.58	0.85 ± 0.79
Carolina buckthorn	0.02 ± 0.02	0.00 ± 0.00	0.09 ± 0.09	0.00 ± 0.00	<0.01	<0.01	0.21 ± 0.21	0.00 ± 0.00
Coralberry	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.01 ± 0.01	<0.01	0.00 ± 0.00	0.00 ± 0.00
Deciduous holly	0.00 ± 0.00	<0.01	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Flowering dogwood	0.00 ± 0.00	0.07 ± 0.07	0.30 ± .030	0.11 ± 0.11	0.00 ± 0.00	0.08 ± 0.08	0.12 ± 0.12	0.00 ± 0.00
Fringe tree	0.00 ± 0.00	0.47 ± 0.47	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Fragrant sumac	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	<0.01	0.00 ± 0.00	<0.01	<0.01	0.00 ± 0.00
Greenbriers	0.06b ± 0.06	0.18b ± 0.11	0.02b ± 0.02	0.03 ± 0.03	0.11 ± 0.11	0.07b ± 0.07	0.61a ± 0.14	0.06b ± 0.06
Hawthorns	0.00 ± 0.00	0.00 ± 0.00	0.05 ± 0.05	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Muscadine grape	0.34c ± 0.31	2.33c ± 2.19	9.89a ± 4.34	2.37 ± 1.55	9.99 ± 9.72	4.76bc ± 3.98	33.87a ± 20.05	6.09ab ± 3.65
Other grapes	0.00c ± 0.00	0.00c ± 0.00	0.48ab ± 0.28	0.24 ± 0.24	0.74 ± 0.74	1.25ab ± 1.09	1.16a ± 0.69	0.02bc ± 0.02
Partridgeberry	0.01a ± 0.01	0.00b ± 0.00	0.00b ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00b ± 0.00	0.00b ± 0.00	0.00b ± 0.00
Poison ivy	0.01ab ± 0.01	0.05b ± 0.03	0.13a ± 0.08	0.04 ± 0.02	0.00 ± 0.00	0.03ab ± 0.02	0.08ab ± 0.07	0.92ab ± 0.82
Pokeberry	0.00b ± 0.00	0.00b ± 0.00	0.54b ± 0.54	0.00 ± 0.00	0.22 ± 0.22	0.05b ± 0.05	0.00b ± 0.00	0.86a ± 0.71
Rattan	0.00 ± 0.00	0.00 ± 0.00	0.06 ± 0.06	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Sassafras	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	<0.01
Smooth sumac	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.65 ± 0.65
Sparkleberry	0.01bc ± 0.01	0.23bc ± 0.23	0.02bc ± 0.02	0.02 ± 0.02	0.12 ± 0.10	0.05b ± 0.03	0.00c ± 0.00	0.28a ± 0.23
Wild plums	0.00b ± 0.00	0.00b ± 0.00	0.06ab ± 0.06	0.01 ± 0.01	0.08 ± 0.08	0.04a ± 0.03	0.00b ± 0.00	0.23ab ± 0.23
Wild rose	0.00b ± 0.00	0.00b ± 0.00	0.00b ± 0.00	0.01 ± 0.01	0.40 ± 0.40	0.07a ± 0.06	0.45a ± 0.30	<0.01ab
Winged sumac	0.00c ± 0.00	0.00c ± 0.00	0.00c ± 0.00	0.00 ± 0.00	3.43 ± 3.43	0.58c ± 0.58	11.06a ± 7.41	22.07b ± 12.82

[a] Within rows, means followed by the same letter were not different ($P < 0.10$) using ANOVA on ranks and Duncan's MRT.

[b] Averaged across all harvested stands which contained greenbelt plots ($N = 16$).

[c] Only combined values were included in the ANOVA.

Table 8—Comparison of mean soft-mast species richness (number of species actually producing soft mast), diversity of production, and evenness among five silvicultural treatments during the first (1994), third (1996), and fifth (1998) year after initial harvest in the Ouachita Mountains

Measure	Year	Unharvested	Single-tree selection	Group selection	Shelterwood	Clearcut
Richness	1994	3.75a[a] ± 0.25	0.50b ± 0.29	1.50b ± 0.65	1.25b ± 0.63	0.50b ± 0.29
	1996	3.75 ± 0.25	6.50 ± 0.87	6.00 ± 1.08	7.25 ± 1.03	5.00 ± 1.08
	1998	4.00a ± 0.41	7.00b ± 0.41	7.00b ± 1.22	8.75b ± 1.38	7.75b ± 1.03
Diversity	1994	0.87a ± 0.15	0.00b ± 0.00	0.16b ± 0.15	0.16b ± 0.16	0.00b ± 0.00
	1996	0.78 ± 0.23	0.83 ± 0.21	0.76 ± 0.12	1.10 ± 0.05	0.67 ± 0.25
	1998	0.70 ± 0.21	0.94 ± 0.18	0.74 ± 0.22	0.96 ± 0.24	0.84 ± 0.17
Evenness	1994	0.54a ± 0.19	0.15b ± 0.14	0.64a ± 0.10	0.15b ± 0.15	0.00b ± 0.00
	1996	0.34 ± 0.18	0.46 ± 0.06	0.53 ± 0.18	0.39 ± 0.15	0.45 ± 0.12
	1998	0.40 ± 0.13	0.50 ± 0.08	0.46 ± 0.12	0.44 ± 0.16	0.38 ± 0.03

[a] Within rows, means followed by the same letter were not different ($P < 0.10$) using ANOVA and Duncan's MRT.

Table 9—-Percent of total soft-mast production represented by pokeberry, blackberry, blueberry, winged sumac, American beautyberry, muscadine grape, and other grapes during first (1994), third (1996), and fifth (1998) year after initial harvest in five silvicultural treatments and unharvested greenbelts in the Ouachita Mountains

Year	Unharvested	Greenbelts	Single tree selection	Group selection	Shelterwood	Clearcut
1994	48	79	49	87	100	100
1996	76	91	86	89	94	99
1998	90	73	95	99	99	95

Pokeberry

In the first postharvest year (1994), pokeberry coverage was greatest in clearcuts and shelterwoods (table 2). Pokeberry plants were not abundant in other treatments and were not detected in unharvested stands. Pokeberry production was greatest in shelterwoods and clearcuts in the first year, and this species produced more soft mast than any other species (table 3). In the third postharvest year, pokeberry coverage was greater in harvested stands than in unharvested stands and greenbelts (table 4), but production had declined. Pokeberry production was greatest in clearcuts, shelterwoods, and group openings (table 5). By the fifth postharvest year, pokeberry coverage was low in all stands, although clearcuts had the highest abundance (table 6). Production in the fifth year declined dramatically, being highest in clearcuts (table 7).

Pokeberry is probably the most important soft mast producer immediately after intensive logging because of its ability to establish quickly and produce abundant levels of soft mast. Pokeberry, a colonizing herbaceous species, invaded the more intensely harvested areas (clearcuts, shelterwoods, and group openings) immediately after harvest, but quickly disappeared because of its inability to compete with the intense woody vegetation growth. This trend suggests pokeberry production reaches a peak about 1-3 years after harvest and then declines quickly.

Blackberry, Winged Sumac, and American Beautyberry

Blackberry plants were present in all treatments the first year (table 2), but produced no soft mast. By the third year, coverage was greatest in harvested stands (table 4). Production was greatest in shelterwoods; no blackberries were produced in unharvested stands and production was almost nonexistent in greenbelts (table 5). By the fifth year, blackberry coverage and production was greatest in clearcuts, shelterwoods, and group openings (tables 6 and 7). Blackberry was the most abundant soft mast in intensively harvested areas the fifth year.

In the first postharvest year, winged sumac coverage was greatest in group selections, shelterwoods, and clearcuts (table 2); however, no production occurred in any stands. In the third year, coverage was greatest in clearcuts, shelterwoods, and group openings (table 4); production was greatest in clearcuts and shelterwoods (table 5). In the fifth year, coverage was greatest in harvested stands (table 6), and production occurred only in clearcuts, shelterwoods, and group openings (table 7).

American beautyberry coverage was low in all stands the first year (table 2), and production was nonexistent. Coverage and production was still low in the third year, and no significant differences in production existed among treatments (tables 4 and 5). By the fifth year, production was greatest in shelterwoods, clearcuts, and group openings (table 7).

Blackberry, winged sumac, and American beautyberry were similar in their response to treatments. Each occurred predominantly in intensively logged areas (clearcuts, shelterwoods, and group openings), and each produced the most soft mast during the fifth postharvest sampling year. These species tended to produce high levels of soft mast when stands were thinned to a residual BA $\leq$11.5 m^2/ha. Blackberry was probably the most important soft mast producer in intensively logged areas by the fifth postharvest year because of its extreme abundance and the high carbohydrate and water content of its mast, which are likely important traits for wildlife during the relatively dry July and early August period.

Muscadine and Other Grapes

Although muscadine grape coverage did not differ among treatments the first postharvest year (table 2), production was greatest in unharvested stands (table 3). In the third year, production was greatest in shelterwoods and single-tree selection stands (table 5). Although no significant difference existed in coverage the fifth year (table 6), production was greatest in shelterwoods and single-tree selection stands (table 7).

For other species of grapes, no difference in coverage or production existed among treatments the first year. In the third year, coverage was greater in single-tree selection stands than in clearcuts or greenbelts (table 4), and production was greater in shelterwoods than in clearcuts or greenbelts (table 5). In the fifth year, production in shelterwoods was greater than in clearcuts, greenbelts, or unharvested stands (table 7).

Initial logging disturbance greatly affected muscadine and other grape species the first year after harvest. However, by the fifth year, grapes (primarily muscadine) were abundant in thinned stands with some intact overstory (shelterwoods, single-tree selections, group openings), but were not abundant in areas with no overstory (clearcuts) or areas with closed canopies and low ground-level light penetration (unharvested areas and greenbelts). However, differences in site, seedbed, previous land uses, or other factors unrelated to treatment may have affected muscadine abundance. For example, among the group selection stands in 1998, muscadine grape covered 23.8 percent of the ground in the north-zone stand (C1124 S11), but was absent in the south-zone stand (C35 S42). Among clearcuts, the north-zone stand (C458 S16) had 19.8 percent muscadine ground cover, whereas the west-zone stand (C1292 S2) had less than 1.3 percent cover.

Blueberries

Blueberry coverage did not differ among treatments any year. However, production was greatest in unharvested stands the first year (table 3); no differences existed in subsequent years. Logging disturbance appeared to reduce production in harvested stands the first year, but production in harvested stands increased in years three and five to levels equal to unharvested stands.

CONCLUSIONS

The importance of individual mast species varied by treatment and years since harvest. Pokeberry was the most abundant mast produced in intensely harvested areas (clearcuts, shelterwoods, and group openings) the first year after harvest, but its production quickly declined in subsequent years. Pokeberry was the only abundant mast-producing herbaceous species on study areas. Herbaceous species tend to recover and establish quickly, whereas woody species tend to recover and establish more slowly. By the fifth year, blackberry became the dominant soft mast available in intensively harvested areas. Winged sumac and muscadine grape were the second and third most abundant mast in these areas by the fifth year. Muscadine grape was generally the most abundant mast in single-tree selections, unharvested stands, the thinned matrix areas of group-selection stands, and greenbelts. Differences in site, seedbed, previous land uses or other factors unrelated to treatment, which were not controlled in this study, may have contributed to the large variance in production and cover we observed among stands of similar treatment.

Providing high-quality, early-successional wildlife habitat is often a priority for land managers. Clearcutting provides abundant soft mast for the first 5 years after harvest, but many people are opposed to this management on public lands. The public generally accepts partial cutting methods more than clearcuts. Our results suggest young shelterwood cuts provide soft mast levels comparable to young clearcuts during the first 5 years of growth. By the fifth postharvest year, total soft mast production in clearcuts and shelterwoods were more than twice the levels of other treatments (Perry and others 1999).

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HOME RANGE, HABITAT SELECTION, AND POPULATION DYNAMICS OF SOUTHERN FLYING SQUIRRELS IN MANAGED FORESTS IN ARKANSAS

James F. Taulman and Kimberly G. Smith[1]

Abstract—After experimental harvests on 18 mature pine-hardwood stands in 6 replicated groups, flying squirrels changed from uniform pre-harvest patterns of nest box use to concentrating on protected greenbelt (GB) areas on harvested stands. Squirrel densities declined on all harvested stands as densities increased on three control stands. Fifty squirrels tracked by radiotelemetry on three control and two harvested stands over three years selected mature pine-hardwood, GB, and hardwood forest habitats for foraging around control study areas; they foraged in GB and adjacent hardwood forests around harvested stands. With boxes closed, squirrels nested in mature pine-hardwood forest on control study areas and GB habitat on harvested stands. Males showed vagility around control areas where mast production was low; female and male ranges were similar, and smaller, around harvested areas where hard mast resources were more abundant.

INTRODUCTION

To effectively consider the needs of wildlife in forest management planning, information on the responses of native species to various disturbance regimes is required (Daily and others 1996, Hurley 1986). Effects of even-aged silviculture on vertebrates, primarily birds, have been documented (see review by Harlow and Van Lear 1987, Smith and Petit 1988). However, few studies have dealt with the autecological responses of mammals to the intermediate disturbances caused by alternative even-aged and selection harvests (Healy and Brooks 1988, Muul and Lim 1978, Nixon and others 1980, Pattanavibool and Edge 1996, Wilson and Johns 1982), and more information is needed (McCoy 1982, 1983; Simberloff and Abele 1976, 1982; Verner 1986; Zimmerman and Bierregaard 1986).

Flying squirrels (*Glaucomys volans*) are common inhabitants of pine-hardwood forests in the southern United States (Goertz and others 1975). Because *G. volans* glides as a primary means of escape and travel, habitat needs include trees of considerable height and a relatively open midstory (Bendel and Gates 1987). Since flying squirrels do not excavate their own tree cavities, den trees and snags are also critical habitat components (Bendel and Gates 1987, Doby 1984, Gilmore and Gates 1985, Muul 1974, Sawyer and Rose 1985, Weigl 1978). Flying squirrels have rather omnivorous food habits but rely heavily on hard mast throughout the year (Harlow and Doyle 1990), e.g. seeds, hickory nuts, and acorns (Braun 1988, Sawyer and Rose 1985, Sealander and Heidt 1990, Sollberger 1943). The life history requirements of flying squirrels suggest the species' utility as a healthy forest indicator for forest-dependent small mammals.

This study was carried out within a larger interdisciplinary research effort in which a range of experimental harvest techniques were evaluated. First, in order to examine the effect of forest alterations on flying squirrel population dynamics, winter nest box surveys were completed on 21 stands (three replicates of each forest condition), providing data on density, fecundity, reproduction, persistence, and overall robustness. Working hypotheses were that none of the demographic parameters investigated would be different either among pretreatment study area groups prior to harvest or among control and harvested study areas during postharvest years.

Second, in order to examine flying squirrel home range and habitat selection behaviors within a fragmented forest landscape, squirrels were tracked by radiotelemetry during late spring and summer on five study areas. Null hypotheses were that male and female home ranges would be similar in size and shape among the five telemetry study areas and that all habitats would be used for nightly foraging in proportion to availability, without selection. It was also assumed that nesting habitats and tree types would be used as available, without selection.

MATERIALS AND METHODS

Population Dynamics

Study areas consisted of 21 15-ha mature pine-hardwood forest stands in the Ouachita National Forest of Arkansas. In summer 1993, 18 of these stands in six replicated groups of three each were harvested to the following specifications: pine-hardwood single-tree selection (STS), pine STS, pine-hardwood shelterwood (SW), pine SW, pine-hardwood seedtree (ST), and pine ST. Detailed descriptions of these habitat characteristics are provided by Baker (1994). During fall 1992, 30 nest boxes were installed on each of the 21 experimental stands in a grid with 60-meter spacing. Boxes were opened in November and nest box surveys were completed once each month during January - March. Boxes were closed for the summer after the last survey. The total number of squirrels found nesting on a study area during the winter survey was used as the population estimate for that stand. Weight, gender, and reproductive condition were recorded for all squirrels.

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Habitat macroplots were installed on the 21 stands in a stratified random array and data were collected by the Silviculture research group in the larger Ecosystem Management research project (see Taulman 1997). Those data are used in this report.

Home Range and Habitat Selection

Telemetry—From 8 - 15 squirrels were radiocollared on each of a subset of five study areas from the total of 21: three control, a pine-hardwood SW, and a pine-hardwood ST. Squirrels were tracked by researchers on foot at night during May - July, 1994 - 1996. One to three independent locations were obtained on each animal on a study area during a 6 - 8 hour tour. Bi-weekly diurnal telemetry led to the discovery of natural nesting locations. After arriving at a squirrel's location, that site was documented by measuring distance and direction to a tree that had been previously monumented using differentially corrected global positioning system (GPS) fixes. Universal Transverse Mercator (UTM) coordinates for each squirrel location were later computed trigonometrically.

Home range computation—Telemetry data sets were examined through location/area curves; those not showing a leveling of area with increasing number of locations were rejected as insufficient to describe an animal's home range. Those data sets showing a range shift during the data collection period were divided and only one segment was used to create a home range estimate. Home ranges were computed using the kernel method, as described by Silverman (1986), Worton (1995), and Seaman and Powell (1996); employing the computer program KERNELHR (Seaman and Powell 1995). The home range used in this report is "the smallest area containing 95 percent of the utilization distribution" (Seaman and Powell 1996). Core activity areas are areas within a contour that contain locations that are closer than would be expected given uniform use of the home range.

Habitat description—For habitat selection analyses of free-ranging squirrels at the five telemetry study areas, additional macroplots were installed and surveyed in habitats frequented by squirrels adjacent to experimental stands. Habitat variables provided a description of woody forest vegetation that was useful in comparisons within and among study areas (Taulman 1997).

Habitat selection analysis—We followed the methods of Manly and others (1993) in describing habitat selection using the resource selection function, or selection ratio, which represents the proportion of a used resource compared to the proportion of that resource available to the animal. Ninety-five percent Bonferroni confidence intervals were constructed for the mean selection ratio for each habitat type on a study area. Where an interval fell completely below 1.00, significant selection against a habitat was indicated; an interval completely above 1.00 indicated significant selection for that habitat type. Available areas were defined individually for each animal and consisted of 99 percent kernel home ranges computed with the so-called "reference" smoothing parameter. This method used with nonnormal data produces contours with inflated areas. This "buffer zone" around the home range provides a useful estimate of habitats available to an individual based on known patterns of habitat use (fig. 1).

Figure 1—Habitat types around study area 27, pine-hardwood shelterwood harvest. This map shows the three contours computed from a telemetry location data set for one individual on which habitat selection analyses were based. Similar contours were created for each of the 50 squirrels in this study for which telemetry location data sets met criteria described in the text. For habitat selection analyses, the outer contour represents available areas for placement of the home range, and is the 99 percent kernel contour computed with the "reference" smoothing parameter. The 95 percent kernel home range estimate computed with least squares cross validation selection of smoothing parameter and the core activity area contours are also shown. Areas of habitats within the 95 percent contour were considered available for the analysis of habitat use within the home range. The core boundary defines an area within which locations were closer together than would be expected based on uniform use of the home range.

Selection was evaluated for placement of the home range within available habitats and for use of habitats within the home range, Johnson's (1980) second and third order selection, respectively. Habitat selection for location of diurnal nesting sites was also investigated (Taulman 1999).

RESULTS

Population Dynamics

While numbers of adult and immature squirrels captured by treatment group increased on control stands from 1993 - 1995, a sharp decline was observed on harvested stands in 1994 (Taulman and others 1998). A partial rebound was seen on some harvested stands during 1995 and 1996;

most of the recovery seen was due to population surges on a few stands that had adjacent hardwood forest habitats. Squirrels on control stands virtually disappeared in 1996 (Taulman and others 1998); most of the squirrels encountered were recaptures from previous years. The ratio of litters to adult females was variable among years and mirrored the change in numbers of litters produced (Taulman 1997), indicating yearly changes in female fecundity.

Nest box use was uniform on all pre-treatment stands, but per capita box use declined on control stands in 1994, as squirrel numbers rose, without a corresponding increase in boxes used (Taulman and others 1998). However, on harvested stands per capita box use increased to a significant level over that seen on control stands, even as squirrel numbers decreased. Comparing nest boxes used to those available, boxes were used significantly more in nonGB habitat on control stands than nonGB on harvested stands in all post-harvest years (Taulman and others 1998). Also, after harvest boxes were used significantly more in GB habitat on harvested stands than in cutover nonGB habitat. Squirrel density on the set of 21 mature pine-hardwood stands in 1993 was significantly correlated with snag density ($r_s = 0.83$, $p = 0.006$) (Taulman and others 1998).

Five mortalities showed direct or circumstantial evidence of predation. Four of those involved collars and remains retrieved either from seedtree harvest habitat (two screech owls [*Otis asio*] and two of unknown causes), and one occurred in a forest/clearing ecotone (rat snake, *Elaphe obsoletus*).

Home Range and Habitat Selection

Habitats categorized differently on each study area were significantly different for many of the measured descriptive variables (Taulman 1997). Overall, habitat types categorized similarly were not significantly different for the same set of variables. Home range areas of males were significantly different among study areas (Kruskal-Wallis ANOVA $p = 0.03$); those of females were similar (table 1). Ranges were smallest and most equal among males and females on harvested study areas; the large male ranges were associated with control study areas. Comparisons with other telemetry studies on flying squirrels show similarities to range areas on the harvested study areas (Taulman 1997).

Mature pine-hardwood forest, GB, and adjacent hardwood-rich forests were selected more than expected for foraging; the selection of GB for home range placement was signifi-

Table 1—Kernel 95 percent topographic home range areas, with 1 standard error (SE), of males and females by study area[a]

	Study areas									
	MAT P-H									
Squirrel gender	N	I	N	II	N	III	N	P-H SW	N	P-H ST
Male	2	14.3	6	19.09 (5.65)	3	44.0 (15.16)	5	4.79 (0.82)	2	2.76
Female	6	6.43 (1.57)	8	6.09 (1.87)	2	6.04	9	5.57 (0.81)	1	3.38

The Kruskal-Wallis ANOVA results indicated that female range areas did not differ among study areas. The K-W ANOVA indicated a significant difference among male range areas ($p < 0.05$), but the multiple comparisons test show no significant difference for any pair due to small sample size.

[a] Only squirrels for which number of locations was ≥ 30 are included.

Table 2—Mean selection ratios (with standard error and 95 percent confidence intervals) for young (< 15 years old) and immature (15 - 40 years old) pine plantation habitats, mature pine-hardwood forest, mature hardwood forest, and greenbelt habitats, including data from all squirrels (N) for which those habitats were available[a]

	95 percent home range			95 percent C.I.		Locations in range			95 percent C.I.	
Habitat	Mean	SE	N	Lower	Upper	Mean	SE	N	Lower	Upper
Young plantation	0.401	0.054	31	0.290	0.511[b]	0.337	0.065	28	0.203	0.471[b]
Immature plantation	0.395	0.115	14	0.147	0.643[b]	0.282	0.102	12	0.058	0.506[b]
Mature pine-hardwood	1.119	0.078	33	0.959	1.279	1.060	0.038	29	0.982	1.137
Mature hardwood	1.555	0.301	15	0.910	2.200	1.414	0.286	11	0.776	2.052
Greenbelt	1.549	0.181	19	1.168	1.930[c]	1.283	0.147	27	0.981	1.586

[a] Selection ratio means for 95 percent home range represent the percentage of habitats in 95 percent kernel home ranges (used) compared with the percentage of those habitats available in a larger area. Available areas and home ranges were computed individually for each squirrel. Means for locations in range represent the percentage of locations in a habitat type in the home range (used) compared with percent area of that habitat in the home range (available).

[b] Indicates significant selection against a habitat type (95 percent confidence interval below 1.00).

[c] Indicates significant selection for a habitat type (95 percent confidence interval above 1.00).

cant (table 2). Mature pine-hardwood forest was significantly selected for diurnal nesting on control study areas; nests were concentrated in GB habitat on the two harvested stands (Taulman 1999). Maps of core activity areas graphically illustrate habitat use and avoidance patterns at all five study areas (Taulman 1997).

The densities of overstory oaks and hickories can provide an indication of the mast production potential in a habitat (Goodrum and others 1971, Wolff 1996). Overstory oak and hickory densities in habitats on these study areas, and fall mast surveys on experimental stands, showed that control study areas were depauperate in overstory hardwoods and mast production (Taulman 1997). The two harvested study areas had habitats containing high densities of overstory hardwoods within or adjacent to experimental stands.

DISCUSSION

Population declines following harvest have been previously documented for flying squirrels (Hokkanen and others 1982) and other vertebrates (Burgess 1971, Wilson and Johns 1982). The marked declines observed on control stands in 1996, and the significantly lower weights of squirrels in 1996, may indicate a metapopulation response to low mast production in 1994 and 1995 (Taulman 1997). Wolff (1996) found that over 14 years population densities of three rodent species varied in synchrony with hard mast production.

Male vagility in areas with limited food resources may be an evolutionary strategy, increasing survival chances of females and offspring kin in familiar nesting areas through reduced competition. Males on distant forays also increase their own likelihood of encountering new females and producing additional offspring. Female philopatry conserves energy and permits young to learn the locations of nearby feeding sites and refugia.

Contrary to the suggestion of some foresters and landscape ecologists (Urban and others 1987) that clearcut harvests mimic natural disturbance events, clearcut silviculture has produced a pattern in which undisturbed forests are now found within the matrix of a fragmented landscape (DellaSala and others 1995, Moloney and Levin 1996), the reverse of the pattern historically observed under natural disturbance regimes (Agee 1994). To ensure the persistence of native terrestrial forest-dependent wildlife, public forests should be managed from a landscape perspective, preserving areas of mature undisturbed forest, with interconnecting corridors of forest habitat (Bright 1993, Dunstan and Fox 1996).

Pickett and Thompson (1978) have suggested the “minimum dynamic area” method as an approach to managing a fragmented ecosystem for the persistence of given species. This method emphasizes landscape management at a scale that can ensure the protection of “recolonization sources” in order to prevent the isolation of small patches and to reduce the possibility of long-term local extinctions. Henderson and others (1985) found that eastern chipmunks (*Tamias striatus*) inhabiting isolated woodlots amid an “ecologically hostile” agricultural landscape were able to disperse rapidly along brushy fencerows to repopulate wooded areas which had undergone local extinctions.

Results of the present study suggest that the U.S. Forest Service compartment level (approximately 10 - 20 sq km) should be an appropriate scale at which to manage habitat for flying squirrels. Within each management compartment many 15 - 40 ha patches of mature pine-hardwood forest, as well as north-slope hardwood forests, should be maintained along with interconnecting corridors through harvested habitats to other undisturbed areas. Some overstory hardwoods and snags should be retained within harvested areas. Provided adjacent mature pine-hardwood, or hardwood, forests are available, and protected GB areas are retained along ephemeral drainages within a treated stand, harvests as severe as the shelterwood prescription should not adversely affect flying squirrel populations. The seedtree harvest regime, even with retained hardwoods, produces a reduction of forest cover which greatly enhances opportunities for such predators as screech owls. The higher predation risk, together with the diminution of mast and cavity resources created by the seedtree harvest regime, appear to render forests subjected to this prescription unsuitable for the long-term persistence of flying squirrels.

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Session 3
Phase II Arthropod and Microbial Diversity Research

David Hammond, Moderator

THE INSECT GUILD OF WHITE OAK ACORNS: ITS EFFECT ON MAST QUALITY IN THE OZARK AND OUACHITA NATIONAL FORESTS

Alex C. Mangini and Roger W. Perry[1]

Abstract—Hardwood regeneration, especially of oaks, is an essential component of ecosystem management in the Ouachita and Ozark Mountains of Arkansas. In addition, oak mast is an important wildlife food. Several species of insects inhabit and consume acorns. Data on the insect guild inhabiting white oak (*Quercus alba* L.) acorns were collected from two undisturbed mature (control) and two single-tree selection stands in the Ouachita and Ozark National Forests from 1993-1998. Insects collected were: weevils of the genus *Curculio* and *Conotrachelus* spp. (Coleoptera: Curculionidae); filbertworm, *Melissopus latifereanus* (Walsingham) (Lepidoptera: Tortricidae); acorn moth, *Valentinia glandulella* Riley (Lepidoptera: Blastobasidae), cynipid gall wasps (stone galls) (Hymenoptera: Cynipidae) and midge larvae (Diptera). Over the 5 years of study, sound acorns averaged 31.9 percent. *Curculio* weevils, with an average infestation rate of 26.6 percent, were the most abundant acorn-infesting insect. Other insect species occurred in much smaller numbers.

INTRODUCTION

Oak regeneration is essential to the health and sustainability of hardwood and mixed pine-hardwood forests in the Ouachita and Ozark Mountains of Arkansas. Also, hard mast produced by oaks is an important food for wildlife in these ecosystems (e.g. Goodrum and others 1971, McShea and Schwede 1993, Nixon and others 1975). Estimates of the effect insects have on acorn quality can help managers assess regeneration potential and hard mast availability.

The insect guild that utilizes acorns includes species in the orders Coleoptera, Lepidoptera, Diptera, and Hymenoptera. Damage to the acorns in most cases results from larvae feeding on the nutritious endosperm of the acorn. Primary acorn-infesting insects are weevils in the genus *Curculio* (Coleoptera: Curculionidae) and the filbertworm, *Melissopus latiferreanus* (Walsingham) (Lepidoptera: Tortricidae). These primary insects infest developing acorns on the tree and are able to directly enter the acorn (Myers 1978). They can completely consume the endosperm, thus destroying the acorn (Gibson 1972). Additional, though less abundant, primary acorn insects include several species of cynipid gall wasps whose larvae inhabit galls on or in the acorn (Myers 1978).

Secondary insects include the acorn moth, *Valentinia glandulella* Riley (Lepidoptera: Blastobasidae), weevils of the genus *Conotrachelus* (Coleoptera: Curculionidae) and Diptera (midge larvae). These insects are scavengers that typically enter acorns which are damaged or have been infested by primary insects (Myers 1978), although Galford (1986) demonstrated that *V. glandulella* can be a primary invader of germinating acorns.

MATERIALS AND METHODS

Our study was part of the USDA Forest Service Ouachita Mountain Ecosystem Management Research Project (Phase II), located in the Ouachita and Ozark National Forests of Arkansas and Oklahoma. This long-term study examines the effects of different timber harvest methods on numerous aspects of forest biology (Baker 1994).

We sampled white oak (*Quercus alba* L.) acorns from four forest stands, yearly, from 1993 to 1998. However, an almost complete mast failure combined with extreme predation by wildlife resulted in few acorns collected in 1998. Therefore, we did not include 1998 data in our analyses.

We selected a single-tree selection (STS) stand and an undisturbed mature forest stand (control) in both the northern and southern portions of the Ouachita region. Pine-hardwood STS stands (C70 S10 and C1649 S13) were initially harvested the summer of 1993 with residual hardwood basal areas (BA) ranging from 2.8 to 5.1 m^2 per ha and total BA ranging from 15.4 to 16.1 m^2 per ha (Baker 1994). Control stands were approximately 80 years old, previously unmanaged, with approximate hardwood BA of 7.1 m^2 per ha and total BA of 29.9 m^2 per ha. In the north zone we used Phase II control stand C284 S11, however, a complete mast failure occurred in this stand in 1994 and we used an alternative stand (C406 S1) as a control that year. Because of low numbers of co-dominant white oaks in the south zone Phase II stand, we selected an alternative control stand (C1652 S58) for sampling in 1993. However, harvesting adjacent to this stand made it unsuitable in 1994. Therefore, from 1994-1998 we sampled an alternative south-zone control stand (C1637 S4).

In each stand, we selected 12-25 mast-producing white oaks and placed 1-6 traps under each tree. Traps were placed half-way between the trunk and canopy edge and in random orientation to the trunk. We used two types of traps: metal trash cans (37 cm tall with a 0.33 m diameter opening) and wooden peach baskets (30 cm tall with a 0.43 m opening). All traps had poultry wire covers to prevent wildlife from removing acorns. In 1993 and 1994 we also collected acorns from a single 1-m radius ground plot encircling each trap. All collections were made in 14-day intervals from late August until all acorns had fallen (late November).

Collected acorns were sorted by stage of development (Myers 1978). Stage 1 and 2 acorns were dead and hard-

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ened fruits aborted soon after pollination. Stage 3 acorns were either green acorns not yet fully developed or brown acorns, apparently mature, but dwarfed. Stage 4 acorns were fully developed but green. Stage 5 acorns were fully mature.

Each acorn was dissected, examined, and tallied as either sound or damaged. Acorn quality categories were: sound acorn, *Curculio*, *Conotrachelus*, filbertworm, acorn moth, Cynipidae, Diptera (midge), rancid, aborted, and wildlife damaged (Myers 1978). The "rancid" category included acorns soured by moisture and/or fungi. If more than one category or damage was present, the acorn was categorized by the primary invader or cause of damage.

For our developmental stage analysis, we pooled data from all four stands and averaged yearly percentages for the five years of data. We included both ground-and trap-collected acorns for 1993 and 1994. For our acorn quality measures, we included only stage 4 and 5 acorns; we excluded data from ground-collections in the analysis of acorn quality because corresponding data were not available for 1995 to 1997. We pooled data from single-tree selection and control stands and calculated yearly percent damage rates to compare damage among groups by year. For overall acorn quality measures, we averaged yearly damage rates for the five years data was collected.

RESULTS AND DISCUSSION

Stage of Development

Fully mature stage 5 acorns constituted the greatest portion of acorns collected (fig. 1). A great portion (34 percent) of the acorn crop was aborts (Stages 1-3). However, because traps were placed in mid-late August of each year, just prior to mature acorn fall, we probably underestimated percentages of aborted and immature.

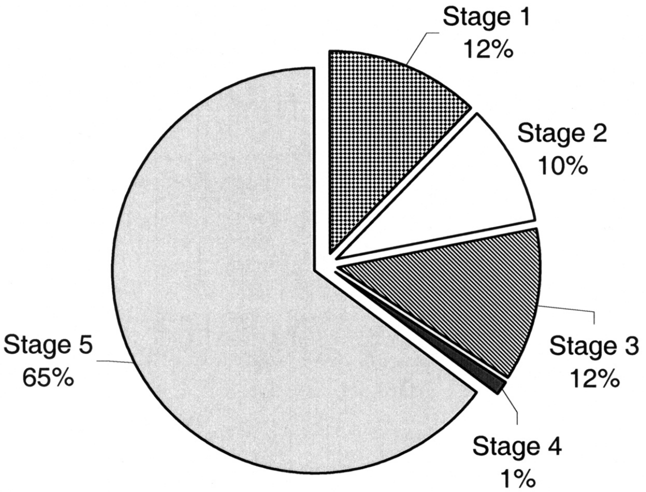

Figure 1—Average proportion of acorns by development stages collected from white oaks located in two single-tree selection and two unharvested mature forest stands in the Ouachita and Ozark National Forests, Arkansas. Percentages were averaged from yearly data from 1993 to 1997. Data for 1993 and 1994 include acorns collected from ground plots and traps; 1995-1997 data include only trap-collected acorns.

Acorn Quality by Year

The white oak acorn crop varied yearly (fig. 2); such variation is characteristic of oaks, with very large crops produced during "mast" years (e.g. Auchmoody and others 1993, Koenig and others 1994). Although more acorns were collected in 1995 than in 1996, the total number of sound acorns was greater in 1996. Total percent soundness was also greater in 1996 than in 1995 (fig. 3). This suggests 1996 may have been a "true" mast year because a greater amount of sound acorns were available for wildlife and regeneration.

Curculio weevils were the most abundant insect infesting white oak acorns, with a substantial presence each year. However, in 1995 and 1996, the *Curculio* infestation rate was lower than in 1994 despite the larger acorn crop of these latter years. This suggests that, in years of abundant mast, the reproductive capacity of this insect is overwhelmed. The other primary insect species, the filbertworm, maintained a smaller but more consistent percentage across years.

Secondary insect species were less abundant than the primary infestors. Percent of acorns damaged by Cynipidae was noticeably greater in 1993, 1995 and 1997 compared

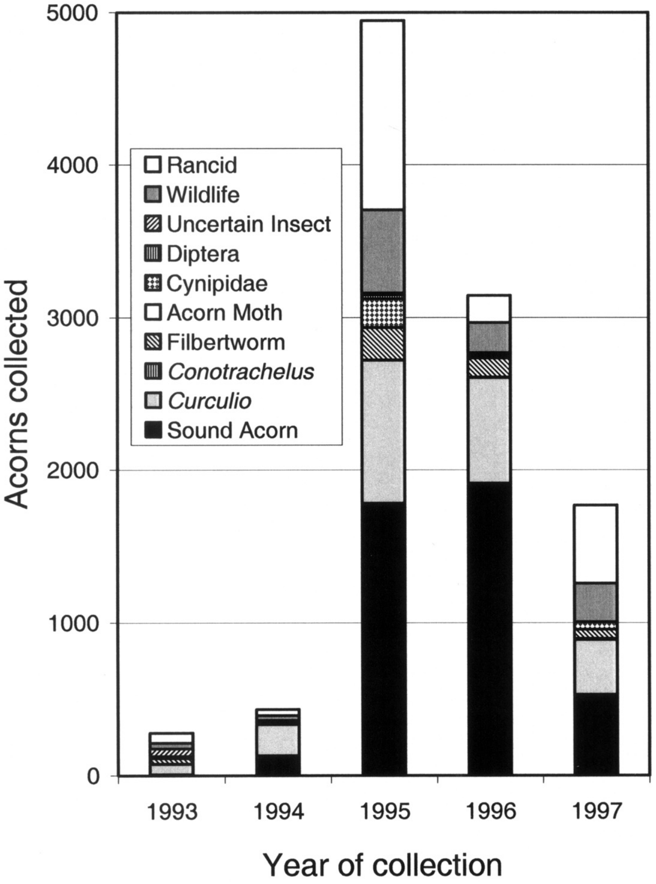

Figure 2—Numbers of white oak acorns, by damage category and year, collected from traps from 1993 to 1997 in the Ouachita and Ozark National Forests, Arkansas. Acorns from two single-tree selection and two unharvested mature forest stands were combined.

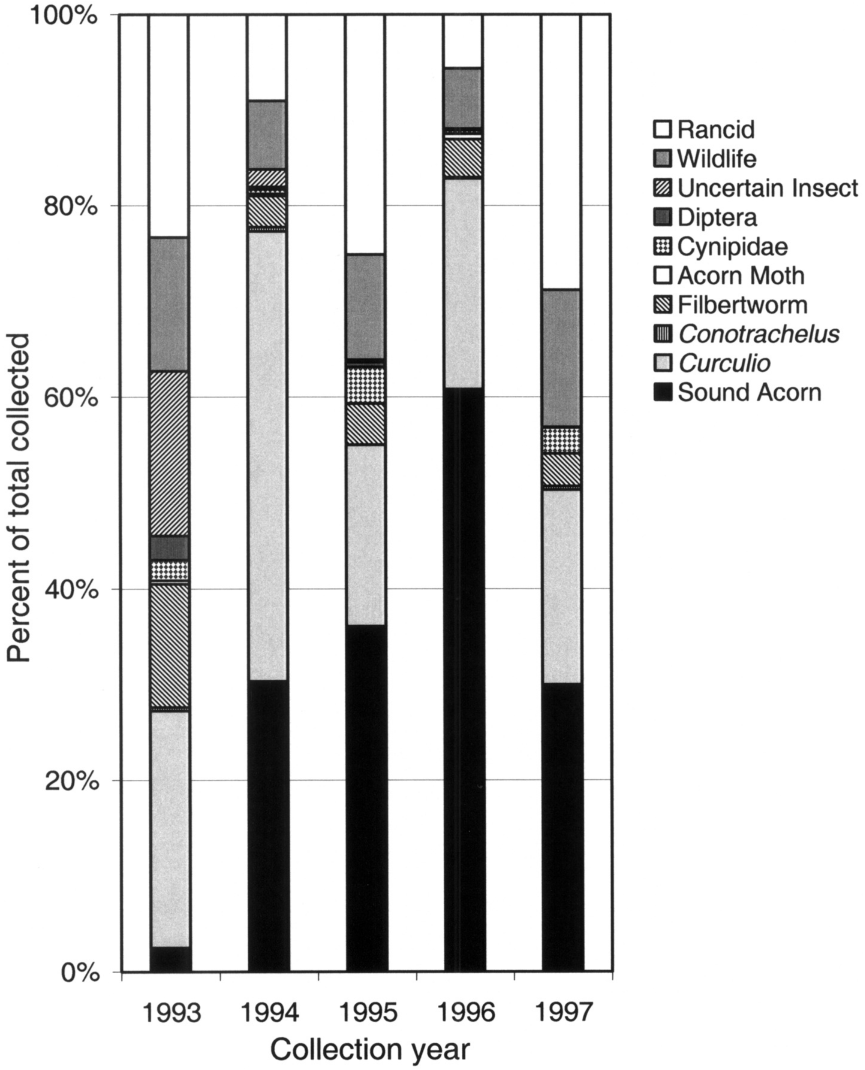

Figure 3—Percentage of white oak acorns in each quality category, by year, collected from traps from 1993 to 1997 in the Ouachita and Ozark National Forests, Arkansas. Acorns from two single-tree selection and two unharvested mature forest stands were combined.

with 1994 and 1996. This suggests a periodicity in Cynipid populations. Acorn destruction by wildlife was fairly consistent across years. However, the damage rate in 1996 was relatively lower than other years suggesting the mast production of 1996 was greater than the consumption rate of wildlife, furthermore suggesting 1996 was a mast year.

Overall Acorn Quality

When averaged over the five years of collection (fig. 4), sound acorns were the largest quality category. *Curculio* weevils were by far the most abundant insects infesting the acorns and surpassed wildlife in numbers of acorns destroyed. However, many *Curculio*-infested acorns had only a few small larvae and these acorns could still be consumed by wildlife and would probably still germinate.

Other insect species were present in much smaller numbers. The acorn moth and *Conotrachelus* weevils were relatively rare inhabitants in acorns. Galford (1986) indicated “secondary” insects such as the acorn moth may destroy significant numbers of acorns that have fallen to the ground and commenced germination. However, because we used acorns collected from traps rather than the ground, the damage rates for these secondary insects may be underrepresented. Perry and Mangini (1997) indicated differences in damage rate estimates for some secondary infestors between ground- and trap-collected acorns.

CONCLUSIONS

Insects feeding on white oak acorns had a significant impact on the quality of the acorn crop. *Curculio* weevils were responsible for the greatest amount of the damage. Curculio infestation, along with wildlife damage and fungi damaged 56 percent of the total mature acorn crop available for regeneration. In addition, a great portion of the overall acorn crop was aborted or fell prior to maturity.

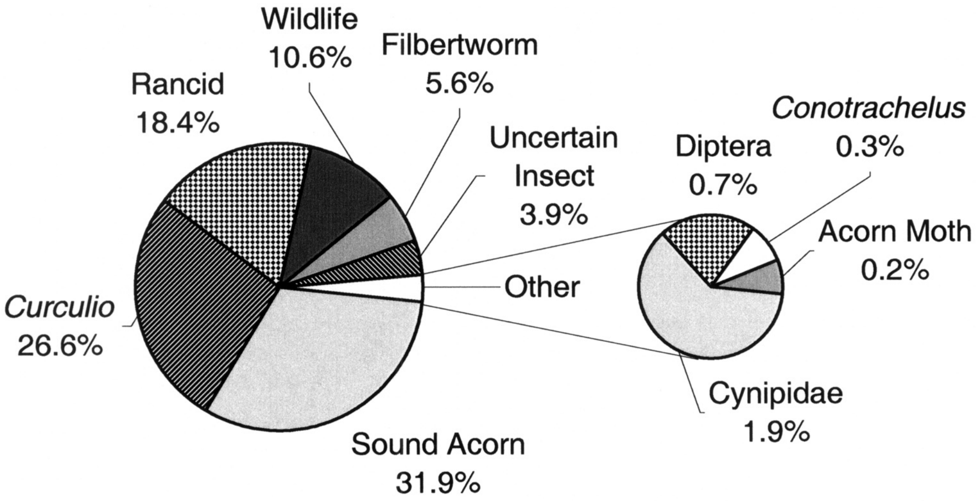

Figure 4—Average percent of white oak acorns for each quality category, collected from traps in the Ouachita and Ozark National Forests, Arkansas. Percentages were averaged from yearly data collected from 1993 to 1997 and represent acorns collected from two single-tree selection and two unharvested mature forest stands.

Our study involved acorns collected during the fall acorn drop. Consequently, our estimates of insect damage are likely to be conservative in regards to regeneration potential because additional infestation and rot could be expected as acorns lay for an extended period on the forest floor prior to sprouting. Furthermore, our overall damage rate estimates were probably reduced because we estimated acorn quality using only trap-collected acorns.

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IMPACT OF STAND MANAGEMENT PRACTICES ON BEETLE DIVERSITY

Stephen P. Cook[1]

Abstract—Insects are useful indicators of change within ecosystems because of their abundance, richness and functional importance. Stand management practices impact the insect community within a forest. Therefore, the objective of the project is to determine the impact of various stand management practices on the diversity of beetles within selected treatments. Four harvesting treatments and uncut controls were examined. The lowest number of beetle families captured was in the control plots. Diversity within individual families can also vary based upon management practices. For example, the same 6 species of Silphidae were captured in 4 of the 5 treatments but fewer individuals were captured in the control plots compared with the treatments. Similarly, the lowest number of Scarabaeidae captured (species and individuals) was in the uncut control plots. Therefore, disturbing stands through tree harvesting increased the overall diversity of beetle families as well as diversity within select families.

INTRODUCTION

Insects and other arthropods are abundant throughout terrestrial ecosystems (May 1989, Wheeler 1990, Gaston 1991) where they play a role in most ecosystem processes and have a major impact on energy flow through a system (Samways 1994). Because of their abundance, species richness and functional importance, insects are useful indicators of ecosystem change (Rosenberg and others 1986). The ecosystem management project in the Ouachita Mountains is examining a variety of timber harvesting treatments and there were 13 stand management treatments implemented (Baker 1994). The objective of my work has been to examine the diversity and abundance of populations of beetles (Coleoptera) following the implementation of 5 of these treatments (unmanaged controls, single-tree selection cuts, group selection cuts, shelterwood cuts and clearcuts).

In the past, stand management research involving insects has primarily focused on species of economic importance, but stand management practices also impact other insect species found within a forest. Studies that examined environmental impacts and used insects as bioindicators range from examining all insect taxonomic groups (i.e. Stork 1991) to using a single insect family such as the beetle family Cicindelidae (Pearson and Cassola 1992). The choice of taxonomic group(s) to utilize for monitoring should be related to the system under examination and the potential impact of the applied treatments on the group(s). Therefore, species do not need to be rare to be useful as bioindicators of system change. In this study, I examined both the overall community of beetles present in the stands as well as 2 specific families (Silphidae and Scarabaeidae). In these stands, most of the individuals of these 2 families that were captured are carrion-feeding insects. In general, the carrion-feeding guild of insects is dependent upon the presence of small vertebrates in the system, which in turn is regulated in large part by the vegetation.

One reason for choosing to examine the carrion-feeding guild of insects is that there are known populations of the endangered American burying beetle, *Nicrophorus americanus*, in western Arkansas and eastern Oklahoma. These populations appear to be remaining fairly stable and are located near where the ecosystem management project is occurring. In addition to American burying beetle, the community of carrion-feeding insects is a large and diverse group. This feeding guild can be sensitive to environmental changes that impact the carrion resource or that result in the introduction of new members to the guild (Summerlin and others 1984, Porter and Savignano 1990, Stoker and others 1995). The diversity and abundance of the guild, along with its sensitivity to environmental changes make it a good candidate group for examining treatment impact on biodiversity.

METHODS

Treatments

The beetle communities occurring within 20 stands (18 in Ouachita National Forest and 2 in Ozark National Forest) were examined. Each stand was approximately 40 acres in size and treatments were applied in 1993 (Mersmann and others 1994 for a more detailed description). Stands were randomly assigned to treatment categories. Four stands per treatment were examined. The treatments were: 1) unmanaged controls in which no harvesting and no other stand management practices occurred; 2) single-tree selection cuts in which some harvesting occurred but a residual basal area of from 45 – 65 ft^2 per acre was retained; 3) group selection cuts in which harvest openings ranged from 0.1 – 1.0 acres and the pine component outside of the openings was thinned to 70 – 80 ft^2 per acre but no hardwoods were harvested or removed from outside of the group openings; 4) shelterwood cuts were harvested while retaining from 20 – 40 of the largest trees per acre (approximately 30 – 40 ft^2 per acre of basal area); and 5) clearcuts in which all trees were harvested except 2 – 5 ft^2 per acre of hardwoods that were retained as den trees and mast producers. A more detailed description of the treatments can be found in Baker (1994).

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Insect Trapping

The beetle communities present within the stands were examined using 4 types of traps. One malaise trap (7 foot tall) was placed in each of 15 stands (3 stands per treatment) for 3 consecutive days during May and June of 1998. These are unbaited traps that indiscriminately capture flying insects. Insects were removed at the end of the 3-day period. Carrion-baited pitfall traps (Lomolino and others 1995) were placed in 10 stands (2 stands per treatment) during May and June of 1998. An 8-trap transect was used and monitored daily for 3 consecutive days. Transects were placed near the center of each stand and traps were approximately 25 yards apart within the transects. Beetles were removed from the traps daily and the traps were rebaited as needed. A second transect of 8 pitfall traps baited with a combination of EtOH and turpentine were placed in the plots approximately 50 yards from the first transect. These traps were also examined daily for 3 consecutive days. These traps are designed to capture pine regeneration insects (Hunt and Raffa 1989). Lindgren funnel traps (45 inches long with a trap surface area of 538 in^2) were placed in 10 stands (2 per treatment) during May of 1999. There were 2 traps placed in each stand, 1 trap was baited with EtOH and alpha-pinene and the other trap was baited with EtOH, alpha-pinene and frontalin, the aggregation pheromone of the southern pine beetle. These traps were left in place for 3 weeks. These traps capture a wide variety of beetles but the specific baits were chosen to sample for bark beetles and their associates.

Analyses

All of the individual beetles captured in all of the traps were identified to family. The data from all trap types were combined to compare family richness among treatments. Prior sampling had occurred on several of these sites following application of the treatments in 1993 (Carlton and others 1994). The two sampling periods had 3 treatment types in common (uncut controls, group selection cuts and shelterwood cuts). For these 3 treatments, data on beetle families was compared between the sample dates for family richness and shared families.

The individual beetles of 2 families, Silphidae and Scarabaeidae, were identified to genus (and species where possible). Within each of the 2 families, comparisons of total number of beetles captured, species richness and species diversity were made among treatments. For these comparisons, species richness refers to the number of species captured and species diversity was calculated using the Shannon-Weaver diversity index (see Price 1975). The 3 most commonly captured species in each of these 2 families was determined and the number of individuals captured in the carrion-baited pitfall traps compared among treatments. The comparisons were based on the number of beetles captured per trap per night and only undisturbed traps were used in the comparison.

RESULTS AND DISCUSSION

Overall Family Diversity

There were a total of 47 families of beetles captured during the 1998 and 1999 sample periods (table 1). The lowest number of beetle families (n=30) was captured in the unmanaged control stands. The stands that received some harvesting treatment all had a larger number of beetle families present and the number of families captured was similar among stands (n=36 to 39). Therefore, some stand disturbance through harvesting appeared to increase the diversity of beetle families present in these stands.

A similar pattern can be distinguished in the 1993 data presented by Carlton and others (1994). They reported capturing a total of 54 families. Of the 3 treatments in common between the 1993 and 1998/99 samples their control plot also had the lowest number of families captured in 1993 (n=23) followed by the shelterwood cut (n=33) and the group selection cut (n=40).

It is interesting to note that while the sampling for these 2 studies occurred 5 to 6 years apart and different trapping schemes were used, there were 36 families of beetles captured in common between the 2 studies. Further, the lowest number of families was captured in the control stands during both studies. However, 29 families of beetles were unique to one or the other study. This may be largely due to the time difference between when trees were harvested and when beetle sampling occurred. This demonstrates the need to sample over a long period of time to establish patterns in diversity or the impact of disturbance events on insect populations.

Diversity Within the Family Silphidae

Although there are known populations of American burying beetle in western Arkansas and eastern Oklahoma, none were captured on these sites during the 2 summers that sampling was conducted. One explanation is that the beetle seems to do best in areas that are undisturbed and have had little human activity (see Ratcliffe 1996). Also, the study plots are relatively small and the beetle may be present in the National Forest(s), just not in these study plots.

There were 6 species of Silphidae captured in 4 of the 5 treatment types (table 2). In the other treatment type, shelterwood cuts, 5 of the 6 species were captured. The lowest number of individual silphids captured was in the control stands, followed by the shelterwood cut stands, the clearcuts, the single-tree selection cut stands and the group selection cut stands. Although the measures of diversity were similar across treatment types, the lowest diversity (H'=1.21) occurred in the shelterwood cut stands. The measure of diversity was similar in the control stands (H'=1.22) and increased in the stands that had been clearcut (H'=1.31) or had been harvested as single-tree selection cuts (H'=1.43) or group selection cuts (H'=1.45). The 2 most abundant species of silphids captured were always the 2 species of *Oceoptoma*. These 2 species are the smallest of the silphids captured during this study and they are both very active during the spring (see Ratcliffe 1996).

As would be expected, different species demonstrated different responses to the harvesting disturbance. The capture distributions of the 3 most commonly caught silphids are presented in figure 1. For *Oceoptoma inaequale*, there was a slight increase in abundance with increasing harvest as long as some residual canopy remained. There was a pronounced decrease in capture of this beetle in the clearcut stands. In contrast, captures of *O. novaboracense* typically

Table 1—Families of beetles captured within stands that had received 1 of the 5 harvesting treatments

	Treatment				
Family	Control	Single tree	Group selection	Shelterwood	Clearcut
Rhysodidae	x	x	x	x	x
Carabidae	x	x	x	x	x
Hydrophilidae	x	x	x	x	x
Histeridae	x	x	x	x	x
Pselaphidae	x	x	x	x	x
Staphylinidae	x	x	x	x	x
Leiodidae	x	x	x	x	x
Silphidae	x	x	x	x	x
Eucinetidae		x		x	
Scarabaeidae	x	x	x	x	x
Byrrhidae		x			
Ptilodactylidae				x	
Buprestidae	x	x	x	x	x
Elateridae	x	x	x	x	x
Throscidae	x				
Eucnemidae		x	x	x	x
Phengodidae	x	x	x		x
Lampyridae		x			x
Cantharidae	x	x	x	x	x
Lycidae		x	x	x	x
Dermestidae	x	x	x	x	x
Anobiidae	x	x	x	x	x
Bostrichidae		x	x	x	
Lyctidae	x	x	x	x	x
Trogositidae	x	x	x	x	x
Cleridae	x	x	x	x	x
Melyridae		x		x	
Nitidulidae			x	x	x
Cucujidae			x	x	
Eroltylidae		x	x	x	
Phalacridae				x	
Coccinellidae	x	x	x	x	x
Endomychidae					x
Lathridiidae			x	x	x
Colydiidae		x	x		x
Tenebrionidae	x	x	x	x	x
Oedemeridae	x	x	x	x	x
Melandryidae	x	x	x	x	x
Mordellidae	x	x	x	x	x
Pedilidae	x	x	x	x	x
Cerambycidae	x	x	x	x	x
Bruchidae		x	x	x	x
Chrysomelidae	x	x	x	x	x
Anthribidae	x				
Curculionidae	x	x	x	x	x
Platypodidae		x			
Scolytidae	x	x	x	x	x

Table 2—Members of the family Silphidae captured within stands that had received 1 of the 5 harvesting treatments

	Treatment				
Species	Control	Single tree	Group selection	Shelter-wood	Clear-cut
Necrophila americana	1	4	10	5	13
Oiceoptoma inaequale	19	57	53	46	31
O. novaboracense	23	28	63	23	53
Nicrophorus orbicollis	7	14	19	9	1
N. pustulatus	1	7	4	0	1
N. tomentosus	1	10	13	3	15
Total capture	52	120	162	86	114

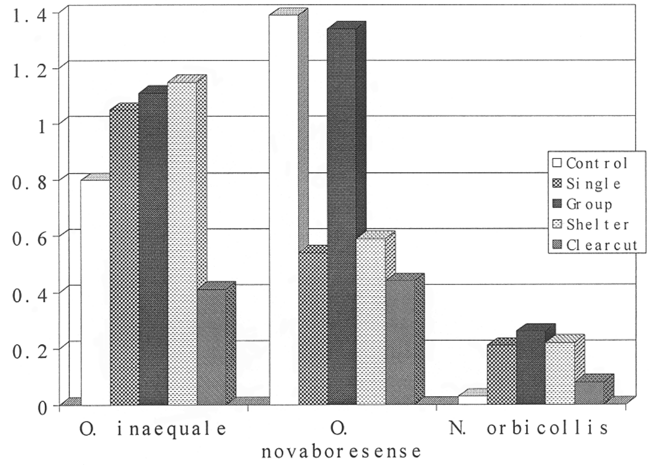

Figure 1—The number of *Oiceoptoma inaequale*, *O. novaboracense,* and *Nicrophorus orbicollis* captured per trap night in the pitfall traps that were baited with carrion. There were eight traps per stand and trapping was conducted over three consecutive nights.

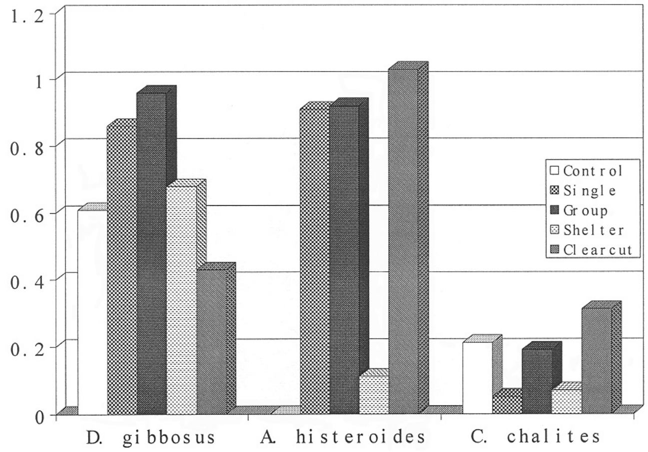

Figure 2—The number of *Deltochilum gibbosus*, *Ateuchus histeroides,* and *Canthon chalcites* captured per trap night in the pitfall traps that were baited with carrion. There were eight traps per stand and trapping was conducted over three consecutive nights.

decreased with increasing tree harvest. Trap catch for the third species, *Nicrophorus orbicollis*, initially increased but then declined as harvest intensity increased beyond the group selection cuts.

Diversity Within the Family Scarabaeidae

There were 20 species of scarabs captured in the stands during the 2 years of this project (table 3). The lowest number of species (n=11) and the fewest individuals (n=67) were captured in the control stands. The largest number of species (n=15) and the most individual scarabs (n=191) were captured in the group selection stands. The measure of species diversity was lowest in the control stands (H'=1.72) followed by the single-tree selection stands (H'=1.93), the clearcut stands (H'=1.94), the shelterwood cut stands (H'=2.04) and the group selection cut stands (H'=2.16).

The 3 most abundant species captured were *Deltochilum gibbosus*, *Ateuchus histeroides* and *Canthon chalcites*. The capture distributions for these 3 species are presented in figure 2. Captures of *D. gibbosus* initially increased with harvest disturbance in the single-tree selection and group selection stands but then decreased in the shelterwood cut and clearcut stands. No *A. histeroides* were captured in the control stands, but the species was present in all of the stands that had received some harvesting treatment. There was no discernable pattern in the distribution of *C. chalcites* captures. As with the silphids, there was a species-specific response by the beetles to harvest disturbance intensity.

CONCLUSIONS

Overall, stand disturbance through tree harvesting increased the diversity of beetle families present in stands. For the individual families examined, species richness was similar among treatments for the Silphidae but the fewest number

Table 3—Members of the family Scarabaeidae captured within stands that had received 1 of the 5 tree harvesting treatments

	Treatment				
Species	Control	Single tree	Group selection	Shelter-wood	Clear-cut
Aphodius sp.	1	1	0	0	0
Ateuchus histeroides	0	40	44	5	38
Canthon sp.	1	0	0	0	0
Canthon chalcites	10	2	9	3	12
Cloeotus sp.	0	0	1	1	4
Deltochilum gibbosus	32	39	45	31	18
Geotrupes blackburnii	6	6	9	4	2
Glaphyrocanthon viridis	0	0	1	2	0
Onthophagus hectate	0	0	16	2	10
O. orpheus	0	1	2	10	4
O. pennsylvanicus	0	15	29	0	2
O. striatulus	4	1	3	0	1
Phyllophaga sp. 1	2	12	9	4	1
P. sp. 2	1	0	0	0	0
P. sp. 3	0	3	4	1	0
Trichiotinus bibens	0	0	0	1	0
Trox sp.	0	4	6	6	5
T. monachus	7	5	12	2	2
T. punctatus	2	0	1	0	1
T. suberosus	1	6	0	3	0
Unidentified	0	0	0	1	0
Total capture	67	135	191	76	100

of silphids were captured in the control plots. The control stands also had the lowest abundance, species richness and species diversity in captured Scarabaeidae and the highest measure for all 3 of these parameters was in the group selection cut plots. As expected, there were species-specific responses to the intensity of tree harvesting disturbance. In general, stand disturbance caused by tree harvesting increased the diversity and abundance of beetles within stands and this could still be measured 5 to 6 years following the tree harvests.

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RADIOGRAPHIC ANALYSIS OF SHORTLEAF PINE SEEDS FROM THE OUACHITA AND OZARK NATIONAL FORESTS

Alex C. Mangini, William W. Bruce, and James L. Hanula[1]

Abstract—Shortleaf pine, *Pinus echinata* Mill., is indigenous to the Ouachita Mountains and the Magazine Mountain area of Arkansas. Natural regeneration of shortleaf pine is a priority on National Forest lands in this area. Insects infesting cones and seeds of shortleaf pine reduce the healthy seeds available for natural regeneration. Two species of seed bug, the leaffooted pine seed bug, *Leptoglossus corculus* (Say) (Hemiptera: Coreidae) and the shieldbacked pine seed bug, *Tetyra bipunctata* (Herrich-Schaffer) (Hemiptera: Pentatomidae) damage pine seeds. We collected cones on or near twenty Ecosystem Management Phase II wildlife sites during late October-early November from 1995 through 1998. In 1998, cone production was too low to include in the analyses. Cones were kiln-dried to open them. Seeds were extracted, dewinged and radiographed. Radiography allows accurate evaluation of healthy and damaged seed. Seed bug damage was consistent with that recorded at the Ouachita Seed Orchard in Mt. Ida, AR. Seed bugs were responsible for about 30 percent damage for combined collections from 1995-97.

INTRODUCTION

Insects feeding on cones and seeds of pines reduce seed and cone production of southern pines. Seed bugs were first recognized as pests of seeds when tree improvement pine seed orchards were established in the South in the late 1960's (DeBarr 1967, DeBarr and Ebel 1974). Two species of seed bugs damage to seeds, the leaffooted pine seed bug, *Leptoglossus corculus* (Say) (Hemiptera: Coreidae) and the shieldbacked pine seed bug, *Tetyra bipunctata* (Herrich-Shaffer) (Hemiptera: Pentatomidae) (DeBarr 1967, Ebel and others 1980). Seed bug damage has been extensive in Arkansas shortleaf pines; for example, at a seed orchard in Mt. Ida, seed bug damage was 43 percent in 1981 (J.C. Weatherby, H.N. Wallace, 1985. Post-treatment evaluations of the 1981, 1982, 1983, and 1984 pest management programs at the Ouachita seed orchard, Mt. Ida, Arkansas. Unpublished report 85-2-10. USDA Forest Service, State and Private Forestry, Southern Region, Alexandria Field Office, Atlanta, GA).

Seed bugs are important considering that natural seed production is highly variable from year to year (Wittwer and Shelton, in press). This becomes all the more important given the desire for natural regeneration on the Ouachita and Ozark National Forests.

In this study, we too estimated the amount of damage caused by seed bugs in natural stands of shortleaf pine on the Ouachita and Ozark National Forests. This is consistent with the Ecosystem Management Phase II goals because seed bugs are endemmic to these areas and we need to estimate their impact in the ecosystem (Mangini and others 1974).

MATERIALS AND METHODS

We collected shortleaf pine cones on or near the twenty Ecosystem Management Phase II Wildlife sites (Logan, Montgomery, Polk, Scott and Yell Counties, AR; LeFlore County, OK) (Baker 1993) during late October-early November from 1995 through 1998. In stands which had been clear-cut, we sampled shortleaf pines adjacent to the site.

Collections were made by shooting cone-bearing branches from the trees using a rifle, either a "243" bolt-action rifle or a "22" magnum rifle. Twenty cones were collected at each site from one or more trees. When cones were sparse at a given site, we collected as many as possible within a 30 minute period.

Cones were placed in paper bags and left in an air-conditioned room for 4-6 weeks to dry. The bags of cones were then placed in a drying kiln at 45°C for 24-36 hours in order to open the cones. Seeds were extracted from the opened cones, dewinged by hand and placed in 7.6 x 12.6 kraft-paper envelopes. These were radiographed to determine the number of healthy and damaged seeds and the causes of damage. Each seed was tallied according to appearance (Bramlett and others 1977, DeBarr 1970) and placed into one of the following seed quality categories: full (healthy), empty, seed bug, abort, seedworm and other.

Data were summarized using standard procedures in SAS (SAS Institute 1989). Means and percentages for each seed quality category were calculated by year and site.

RESULTS AND DISCUSSION

Although we sampled from 1995 through 1998, the 1998 cone crop was extremely sparse so those data are not included in the results. When data from the three years are taken as a whole, we found that 67 percent of the seeds were healthy (fig. 1). Only 4 percent of the damage could be attributed to insects. When the insect damage is further broken down, only 11 percent of the damage was due directly to seed bugs (fig. 2). On the other hand, seedworm caused 38 percent of the insect damage. Three species of

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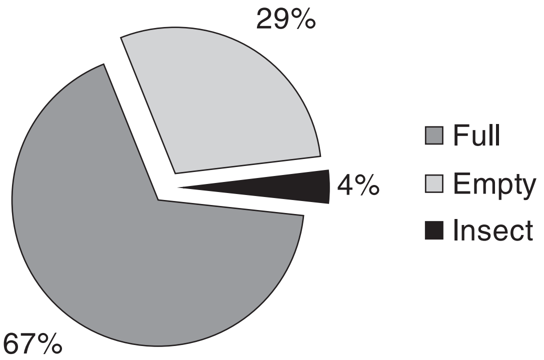

Figure 1—Seed quality as percent of total seeds for shortleaf pine from the 20 Ecosystem Management Phase II wildlife stands on the Ouachita and Ozark National Forests in Arkansas. Data represent combined counts of collections made in 1995 through 1997. Quality determined by radiographic examination.

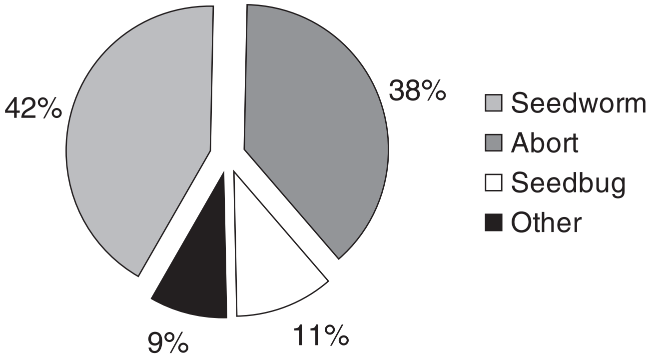

Figure 2—Categorization of “insect” damaged seeds of shortleaf pine collected from the 20 Ecosystem Management Phase II wildlife stands on the Ouachita and Ozark National Forests in Arkansas. Data represent combined counts of collections made in 1995 through 1997. Quality determined by radiographic examination.

seedworm (*Cydia* spp. = *Laspeyresia* spp.) commonly infest seeds of southern pines (Ebel and others 1980).

These results may be misleading if the empty (fig. 1) and aborted (fig. 2) categories are not considered. Radiographs of seeds can clearly show effects of late-season feeding by seed bugs (Bramlett and others 1977); however, earlier feeding on second-year cones or seed bug feeding on first-year cones can cause aborted or empty seeds.

There are two causes of ovule abortion in the first year of cone development: poor pollination and seed bug damage. If pollen is not present when flowers are receptive or if the pollen does not germinate, the ovules abort early in the first year (Bramlett and Johnson 1975). Nymphs of *L. corculus* are known to feed on the flowers of shortleaf pine and destroy ovules (DeBarr and Kormanik 1975, Ebel and Yates 1974); but the damage is difficult to distinguish from those aborted by lack of pollen (Bramlett and others 1977).

Consequently, the empty and abort categories in figures 1 and 2, respectively, when added to the radiographically-confirmed seedbug damage are likely a more accurate estimate of the effects of the two seed bug species. This is consistent with estimated damage in the Ouachita seed orchard from 1981-84 where the proportion of empty seeds ranged from 23-70 percent and the confirmed seed bug damage ranged from 2-43 percent ((J.C. Weatherby, H.N. Wallace, 1985. Post-treatment evaluations of the 1981, 1982, 1983, and 1984 pest management programs at the Ouachita seed orchard, Mt. Ida, Arkansas. Unpublished report 85-2-10. USDA Forest Service, State and Private Forestry, Southern Region, Alexandria Field Office, Atlanta, GA).

Our results are consistent with Shelton and Wittwer (1996) in that the quantity of sound seeds produced is positively related to total seeds produced. In 1996, shortleaf cone production was much greater than that in 1995 and 1997 (fig. 3). Healthy seed was substantially higher in 1996 (fig. 4) at 62.5 percent, near the maximum indicated by Wittwer and Shelton (in press). They demonstrate a positive response of healthy seeds as a function of total seeds produced up to an asymptotic limit at about 60 percent healthy seeds for the largest seed crops.

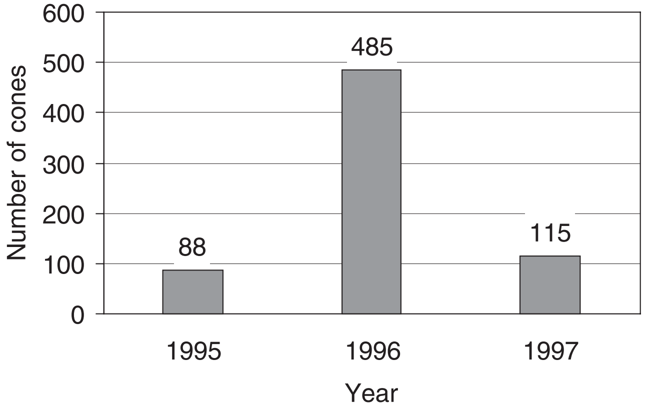

Figure 3—Shortleaf pine cones collected from the 20 Ecosystem Management Phase II wildlife stands on the Ouachita and Ozark National Forests in Arkansas.

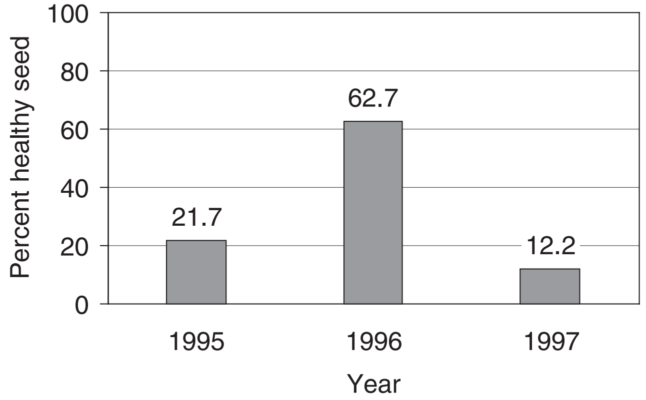

Figure 4—Healthy seeds as a percentage of total seeds collected for shortleaf pine from the 20 Ecosystem Management Phase II wildlife stands on the Ouachita and Ozark National Forests in Arkansas.

It is apparent that managers should monitor the shortleaf seed crop in its first year of development to determine crop estimates. For natural regeneration, site preparation efforts can be timed to coincide with large seed crops. This will also minimize damage due to seed bugs and seedworms.

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CROWN HEALTH OF RESERVE HARDWOOD TREES FOLLOWING REPRODUCTION CUTTING IN THE OUACHITA MOUNTAINS

Dale A. Starkey and James M. Guldin[1]

Abstract—Monitoring the health of reserve hardwood trees is being performed as part of the Ecosystem Management Research Project on the Ouachita and Ozark National Forests in Arkansas. A suite of crown variables (diameter, live crown ratio, density, dieback, and foliage transparency) was used to detect significant changes in reserve tree health over time. While treatments had some effect on crown variables over time, seasonal climatic conditions (e.g. acute drought) may have had a greater effect. It was generally apparent that for the most intensive treatments, crown variables worsened more over time compared to less intensive treatments. Results will provide information about the success of retaining such trees and provide guidelines for selecting reserve trees in future operational harvests.

INTRODUCTION

An important element of the Ecosystem Management Research Project in the Ouachita Mountains is to investigate whether the shortleaf pine (*Pinus echinata*)-hardwood forest type in the Ouachita and Ozark-St. Francis National Forests can be managed using even-aged and uneven-aged reproduction cutting methods that rely on natural regeneration. A complete description of the project can be found in Baker (1994). In about half of the harvest treatments, overstory and midstory hardwoods are being retained in order to meet ecosystem management objectives such as improved wildlife habitat, greater biodiversity, reduced visual impact of harvesting, and perpetuation of the pine-hardwood forest type.

Retained hardwood trees generally meet the definition of “reserve trees” (Helms 1998) for the even-aged methods, since they are retained after the regeneration period. However, in the uneven-aged methods, retained trees are not truly reserve trees; they are more appropriately considered as a component of the pine-hardwood forest type being studied in this project. Nonetheless, we will use the term “reserve trees” throughout to describe hardwoods retained after reproduction cutting in these treatments.

The health and longevity of reserve trees are important if ecosystem management objectives are to be realized. But little information exists about the health and fate of these trees that might guide the selection of reserve trees in future operational harvests under an ecosystem management regime.

The major concern for the health of reserve hardwoods is oak or hardwood decline (Starkey and others 1989, Wargo and others 1983). Decline can generally be described as a complex disease syndrome resulting from the interaction of a variety of host, site, and stand factors with biotic and abiotic agents and stress factors. It is expressed by a progressive dieback of the crown from the upper and outer portions downward, usually resulting in mortality. Manion (1991) describes decline as “an interaction of interchangeable, specifically ordered abiotic and biotic factors to produce a gradual general deterioration, often ending in death of trees”; decline is depicted as a spiral of (1) one or more predisposing factors, followed by (2) one or more inciting factors, which are then followed by (3) one or more contributing factors. Factors that can be responsible for decline include abiotic, biotic, site/stand or anthropogenic agents (fig. 1).

In oak decline, specific predisposing, inciting and contributing factors interact to affect the incidence and severity of symptoms (fig. 2). In the Eastern United States, decline is attributed to climatic events and site/stand factors, defoliation, drought, frost, root rots and borers (Millers and others 1989, Starkey and others 1989). In the Ouachita Mountains, the factors most likely to be operative are site/stand factors (predisposing), stand disturbance from harvesting, or drought (inciting), and root rot/borers (contributing). Oaks (*Quercus* spp.), hickories (*Carya* spp.), and other hardwoods are susceptible to decline. Oaks in the red oak group are usually more frequently and severely affected than white oaks or hickories. Other hardwoods are less affected.

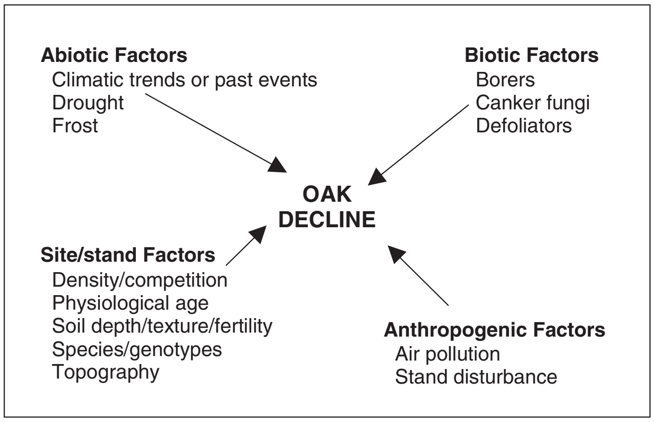

Figure 1—Causal factors of oak decline organized by type.

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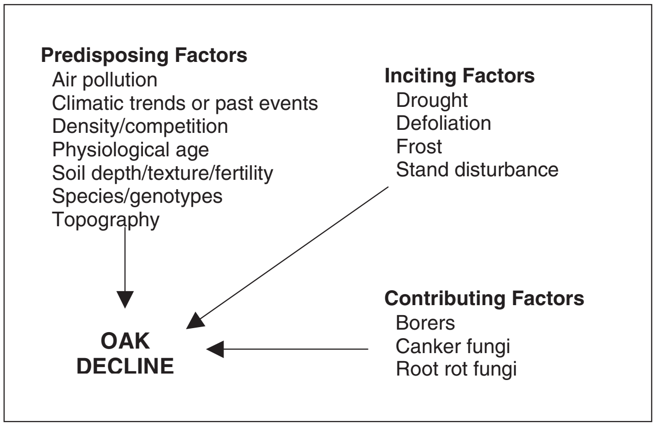

Figure 2—Causal factors of oak decline organized by their function in the decline syndrome.

METHODS

We identified seven treatments in which hardwoods were retained, as follows

1. (1) UC: unmanaged control treatment
2. (2) GSPH: group selection treatment, with hardwood retained within groups
3. (3) STSW: pine-hardwood single-tree selection treatment
4. (4) STSL: low-impact single-tree selection treatment
5. (5) SWW: pine-hardwood shelterwood treatment
6. (6) STPH: pine-hardwood seed tree treatment
7. (7) CC: clearcutting treatment.

Each treatment was installed in 4 stands and measurements were taken on 12 plots per stand for a total of 336 plots in this study. Details about the treatments and sample plot configuration are presented in Guldin and others (1994).

Prior to the imposition of harvest treatments, three to five of the largest hardwoods nearest plot centers were identified for monitoring at each plot. These constituted the trees most likely to be designated as reserves during marking and harvesting operations. Preference was given to trees already marked as reserves, to oaks, hickories, and finally other hardwoods. Azimuth and distance from plot center were recorded for each tree as well as species and diameter at breast height (d.b.h.). Plots were visited and trees evaluated prior to harvest in 1992, 3 years after harvest in 1996, and 5 years after harvest in 1998.

A suite of crown measurements was utilized to evaluate the condition (i.e. health) of tree crowns at each sampling—a procedure currently being used in the National Forest Health Monitoring Program (U.S. Department of Agriculture, Forest Service 1992). The suite consists of six variables (table 1), each requiring two crew members to estimate. Crown diameter was measured by projecting the crown perimeter onto the ground and measuring with a tape; measurements were taken on N-S and E-W axes. The other five variables were each visually estimated by two observers standing on opposite sides of the tree about one-half to one tree length away such that a clear view of the crown was obtained. Estimates for each variable were made by each crew member individually, and a consensus or average of both is used as the final estimate.

Table 1—Crown health indicators for monitoring reserve hardwood trees

Indicator	Definition and units
Crown diameter	Measured on ground in two directions at 90°; in feet to the nearest foot; average of measurements.
Crown position	Standard forestry definitions; dominant, codominant, intermediate, or suppressed.
Live crown ratio	Ratio (in percent) of live crown length to total tree height; visually estimated in 5-percent increments.
Crown density (DEN)	Estimated percentage of foliage, twigs, branches, and reproductive structures blocking light through the crown; visually estimated in 5-percent increments.
Crown dieback (DBK)	Estimated percentage of recent dieback (fine twigs remaining) in upper and outer portions of the crown compared to entire crown; visually estimated in 5-percent increments.
Foliage transparency (TRN)	Estimated percentage of light being transmitted through the foliated portions of the crown; visually estimated in 5-percent increments.

Crown health data were obtained during the leaf-on, summer field season. Field crews were trained at the beginning of the field season to collect crown health data during an all-day session. Classroom training and field practice were followed by field testing and evaluation in order to meet quality assurance goals (U.S. Department of Agriculture, Forest Service 1992). For all visual crown variables, a goal of ±10 percent (i.e., two 5-percent classes, see table 1), 90 percent of the time (when compared to estimates of the trainers) was used. For crown diameter, average diameter was required to be ±10 percent of the trainers result, 90 percent of the time. Catastrophic events (windthrow, logging damage, etc.) occurring to sample trees between visits were identified and these trees removed from the data set used to evaluate the effects of decline.

RESULTS

One thousand and three hardwoods were identified as reserve trees before stand treatments were imposed. Most of these were white oaks (*Q. alba*) (WHO) and post oaks (*Q. stellata)* (PSO); they comprised about 65 percent of the population (fig. 3). Other oaks [black oak (*Q. velutina*) (BLO), southern red oak (*Q. falcate*) (SRO), northern red oak (*Q.*

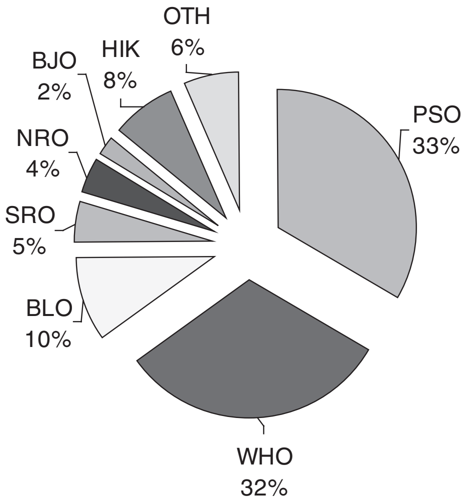

Figure 3—Percent of sample population by species.

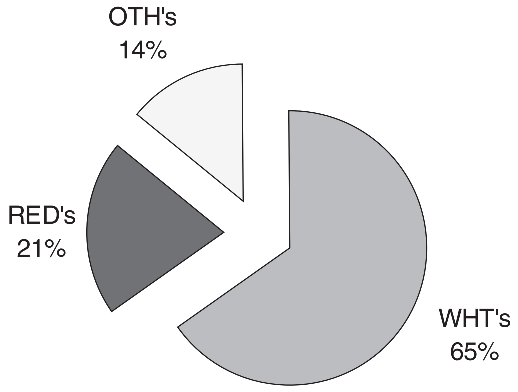

Figure 4—Percent of sample population by species group.

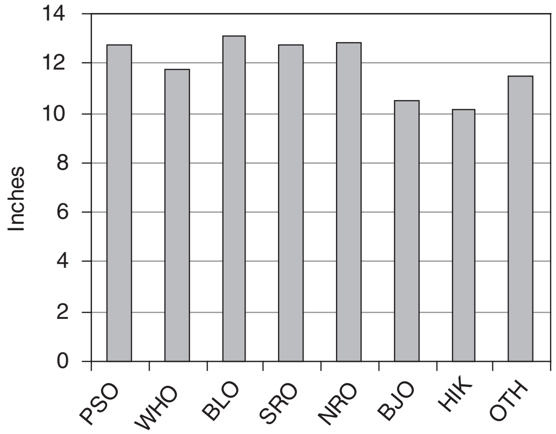

Figure 5—Quadratic mean diameter at breast height for sample population by species.

rubra) (NRO), blackjack oak (*Q. marilandica*) (BJO)]; hickories (*C. texana* and *C. tomentosa*) (HIK); and other hardwoods [sweetgum (*Liquidambar stryracifula)*, blackgum (*Nyssa sylvatica*), winged elm (*Ulmus alata*), black cherry (*Prunus serotina),* white ash (*Fraxinus Americana)*, and sassafras (*Sassafras albidum*)] (OTH) made up the remainder. Since most species were present in relatively small numbers, we used species groupings for all data analysis involving species (white oaks, WHTs include PSO and WHO; red oaks, REDs include the other oaks; and other hardwoods, OTHs including hickories and all other species) (fig. 4).

The d.b.h. of reserve trees varied little among species, ranging from a quadratic mean d.b.h. of 10.1 to 13.1 inches (fig. 5). Most trees were either codominant or intermediate in crown position with the proportion of each varying by species (fig. 6). Generally, 80 to 90+ percent of sample trees were in these two classes. A few dominant or suppressed crown classes were represented in each species tally.

By the 3- and 5-year measurements a number of trees had been cut, damaged or died (fig. 7). By 1998, 876 trees survived in our sample set. Forty-two trees had been cut by

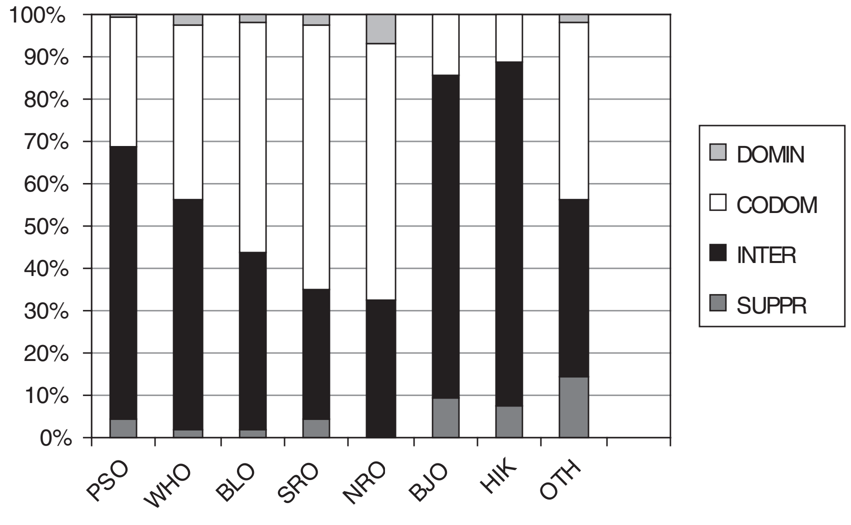

Figure 6—Percent of sample population by crown position and species.

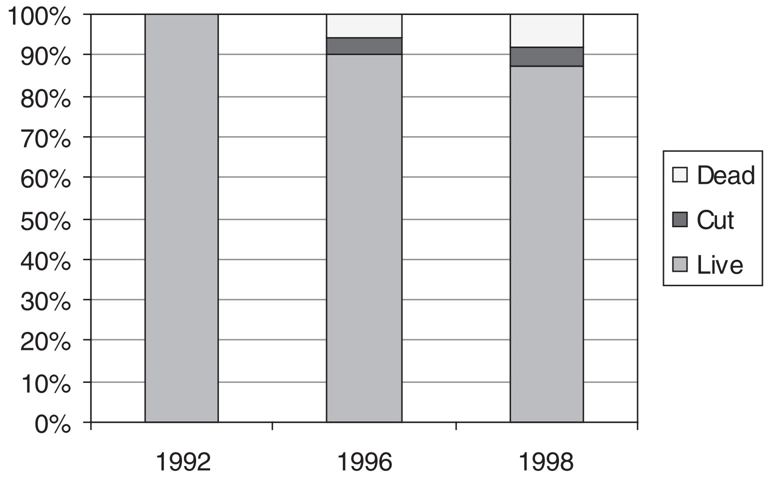

Figure 7—Sample population change due to cutting and mortality.

Table 2—Causes of mortality

Year	Herbicide	Logging	Lightning	Windthrow	I/D/Unknown	Total
1996	1	12	0	2	44	59
1998	0	0	1	1	20	22
Total	1	12	1	3	64	81

1996 and another 4 trees by 1998, either in the initial harvest or subsequent site—preparation treatments. Removing these from consideration, a total of 81 trees died over the 5 years out of 957 trees—a very acceptable survival rate of 91.5 percent. Of the trees that died, a few were attributed to logging, lightning or windthrow, but most were from unknown causes (which could be decline-related) (table 2). If all of the 64 trees which died from unknown causes are attributed to decline, then the mortality rate due to decline after 5 years is 6.7 percent.

Previous experience and research suggests that decline is closely related to inciting events like severe drought (Starkey and others 1989, Wargo and others 1983). Thus, drought can have an effect on reserve tree health in spite of cutting practices imposed. To evaluate this phenomenon we examined the crown variables of trees in the uncut control treatment. Climatic conditions over the 1992-98 period caused a rather large increase in dieback, a decrease in density, and an increase in transparency—all indicative of worsening crown health (fig. 8). The trend is most apparent from 1996-

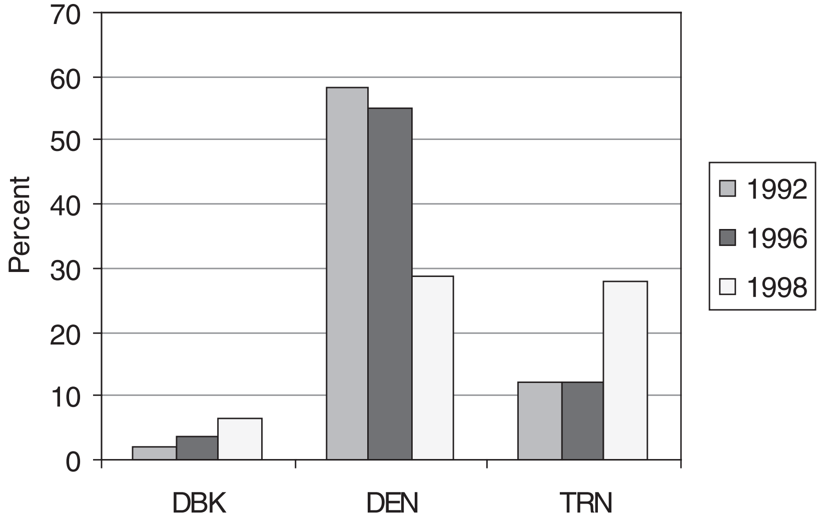

Figure 8—Mean crown variables by year for the uncut control treatment.

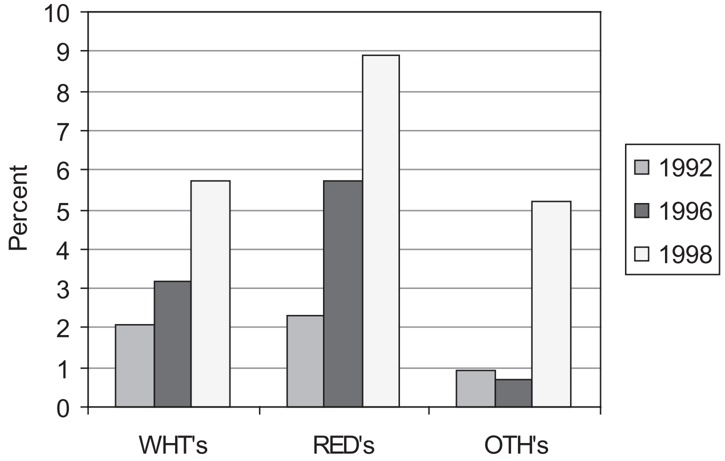

Figure 9—Mean dieback in the uncut control treatment for species groups by year.

98, reflecting the acute drought conditions which prevailed in central and northern Arkansas in the past few years, particularly 1995, 1996, and 1998.

Past experience and research (Starkey and others 1989, Wargo and others 1983) shows that trees in the red oak group are more often and severely affected by decline than trees in the white oak group or other species of hardwoods. This too is evident in the uncut control treatments by examining the crown variables for the species groups. Red oak dieback started out little different from white oaks in 1992, but increased much more in 1996 and 1998 (fig. 9).

Despite the effects of climate, treatment effects are somewhat apparent on reserve tree dieback for all tree species (fig. 10). Dieback increased the greatest amount in 1996 and 1998 in the even-aged treatments—the shelterwood, seed tree, and clearcut—with the greatest increase in the clearcut.

Red oaks began with a little higher dieback levels in 1992 and also seemed to experience higher dieback by 1998, even in the uneven-aged treatments and the uncut control (fig. 11). This again is probably a response to the acute drought conditions that occurred after the treatments were imposed. Nonetheless, 1996 and 1998 dieback was highest in the more intensive treatments.

DISCUSSION

Reserve trees were affected by both climatic variation and by the cutting treatments applied. The largest effect appears to be due to short-term climatic variation, specifically, drought over the measurement period. Survival of reserve trees to date is 91.5 percent, greater than we expected. Although dieback increased over the measurement period, surviving trees had mean dieback levels that are not excessively high (generally < 15 percent) and may not lead to further mortality. We have found in other studies that for oaks, dieback does not generally indicate a high risk for mortality unless it is one-third of the crown or more (Steven W. Oak, Dale A. Starkey [and others]. Unpublished data on file. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Region, Forest Health Protection).

To date, these results suggest that reserve hardwood trees selected from the larger diameter and higher crown classes

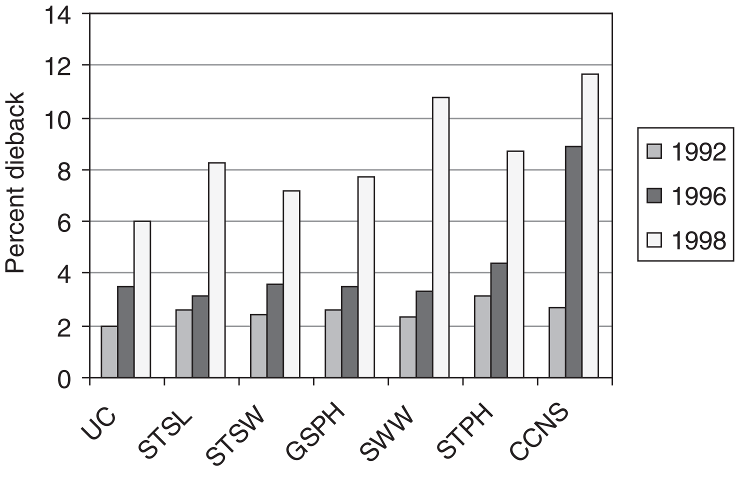

Figure 10—Dieback by treatment and year for all species.

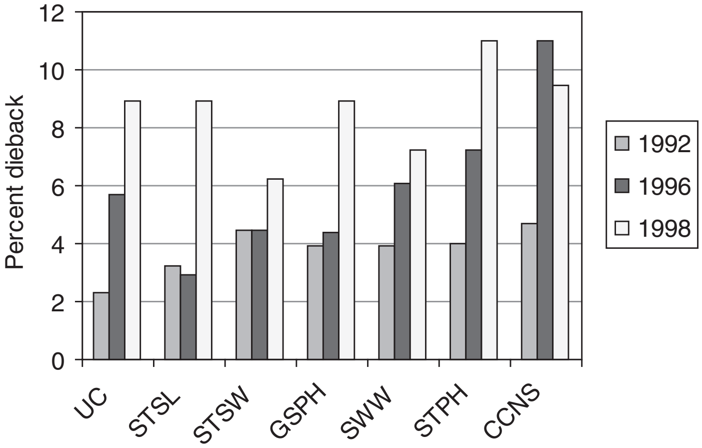

Figure 11—Dieback for red oaks by treatment and year.

can be any species desirable in meeting management objectives. Further, survival and health of reserve trees in this study suggests that reserve tree selection need not be a complicated task, and acceptable results may obtain from simply selecting the largest trees of desired species in numbers that meet the objectives of management.

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Session 4
Phase II Soil and Water Resources Research

Alan Clingenpeel, Moderator

CHANGES IN ORGANIC MATTER AND NUTRIENTS IN FOREST FLOOR AFTER APPLYING SEVERAL REPRODUCTIVE CUTTING METHODS IN SHORTLEAF PINE-HARDWOOD STANDS

Hal O. Liechty and Michael G. Shelton[1]

Abstract—This study was initiated to determine the effects of various regeneration cutting methods on forest floor mass and nutrient content in shortleaf pine-hardwood communities in the Ouachita and Ozark National Forests. Clearcutting generally altered forest floor concentrations of N, P, and S as well as loss on ignition by increasing the amount of herbaceous vegetation and by mixing mineral soil with the forest floor. Reductions in forest floor concentrations of Mg and Mn were evident in regeneration methods that favored pines and excluded hardwoods (pine single-tree selection and pine shelterwoods). Changes were due to the overall lower concentrations of these nutrients in pine foliage than in hardwood foliage and a reduction of Mn in both pine and hardwood foliage after harvesting. Thus reductions in forest floor Mn appeared not only to be related to changes in stand composition but also reduced soil availability and/or tree uptake. Reductions in Mn content after harvesting were between 22 and 52 percent. Reduction of the other nutrients were generally less than 20 percent.

INTRODUCTION

Timber harvesting and associated silvicultural practices has the potential to significantly alter nutrient cycling and organic matter decomposition in forest ecosystems (Yin and others 1989; Montagnini and others 1989; Kimmins 1996; Barnes and others 1998). Tree and vegetation removal reduces annual inputs of litter to the forest floor until such time that aboveground biomass recovers to pre-harvested levels (Kimmins 1996). This reduction in annual inputs is partially offset at the time of harvesting by a substantial one-time flux of woody materials and decomposable tissues added to the soil surface in the form of logging debris and unmerchantable woody material. The reduction of annual inputs and addition of harvesting debris not only impacts the amount but type of organic matter inputs to detritus food chains resulting in the modification of organic matter decomposition and the availability of nutrients in the forest floor and soil. Nutrient mineralization, like decomposition, changes with the modification of organic matter and carbon inputs after harvesting. Removal of forest canopies can also alter nutrient mineralization and decomposition in temperate forests by increasing available water and soil temperature. Increasing soil temperature occurs after harvesting due to the increased insolation after canopy removal (Waide and others 1987; Liechty and others 1992). A reduction in the amount of canopy also reduces transpiration resulting in an increase in soil moisture availability (Liechty and others 1992). Depending on original ambient conditions and the magnitude of change in temperature and moisture, alteration of ambient conditions can either increase microbial efficiency and population levels or decrease microbial activity (Waide and others 1987). Reductions in decomposition or mineralization can also occur if nutrient concentrations are decreased below pre-harvest levels by addition of carbon rich, nutrient poor organic material.

The need to design and develop alternative regeneration strategies for shortleaf pine (*Pinus echinata* Mill.)-hardwood forests in the Ouachita Mountains has stimulated land managers to question how silvicultural practices affect long-term sustainability and productivity. One of the keys to answering this question is a better understanding of the degree to which these strategies alter nutrient cycling and decomposition. We have attempted to evaluate the effect of harvesting and composition manipulation on these processes by quantifying changes in forest floor mass, nutrient contents, nutrient concentrations, and organic matter 3.5 years after application of several regeneration cutting methods as part of the USDA Forest Service's Phase II Ecosystems Management Study. Our objectives were to quantify changes in the: (1) amount and composition of forest floor, and (2) nutrient concentration and content of forest floor. This information will then be used to address if potential long-term changes in nutrient and organic matter in these ecosystems could occur.

METHODS

Study Sites and Design

Relatively undisturbed, mature, shortleaf pine (*Pinus echinata* Mill.)-hardwood stands occurring on generally south facing slopes in the Ouachita Mountains of Arkansas and Oklahoma were selected for study. A detailed description of the study area and its vegetation is provided in Baker (1994) and Guldin and others (1994). Baker (1994) describes the selection criteria for stands and provides an overview of the full array of thirteen overstory treatments that were performed in the summer of 1993. Six of these thirteen treatments were selected for forest floor sampling in this study. The six treatments selected were: clearcut (CC), pine shelterwood (PSW), pine-hardwood shelterwood (PHSW), pine single-tree selection (PSTS), pine-hardwood single-tree selection (PHSTS), and unharvested control (UC). These treatments provided a broad range of disturbance levels but also allowed us to evaluate the effects of different levels of hardwood retention on nutrient cycling. A total of 24 stands, 4 from each of the 6 treatments, were

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Table 1— Mean preharvest (1993) and postharvest (1997) basal area (m^2 ha^{-1}) of conifer and hardwood trees greater than 10 cm d.b.h. for each treatment

	Preharvest		Postharvest	
Treatment	Conifer	Hardwood	Conifer	Hardwood
	- - - - - - - - - - - - - - - *m^2 ha^{-1}* - - - - - - - - - - - - - -			
CC	19.2	6.7	< 0.1	0.9
PSW	19.8	6.7	9.3	1.2
PHSW	25.6	6.4	8.2	3.2
PSTS	20.0	9.7	14.2	1.8
PHSTS	21.6	7.7	11.8	3.1
Uncut	21.5	6.7	20.3	7.0

CC = clearcut; PSW = pine shelterwood; PHSW = pine-hardwood shelterwood; PSTS = pine single-tree selection; PHSTS = pine-hardwood single-tree selection; UC = unharvested control.

used for the study. Each stand was initially subdivided into quarters to facilitate establishment of 12 randomly located, permanent subplots that were used for sampling vegetation. These quarters were oriented perpendicular to the dominant slope within the stand. In stands receiving a uniform manual site-preparation treatment, subplots in one of the four quarters were randomly selected. In stands receiving different site-preparation treatments, forest floor sampling was restricted to the quarter assigned to the manual site-preparation treatment. This assured that the site preparation would be the same in all areas used for forest floor sampling. Subplots in the chosen quarter represented the lower, middle, or upper topographic positions. In total, 72 subplots were sampled from these 24 stands. Pre-harvest and post-harvest conifer and hardwood basal areas in the selected subplots (Guldin and others 1994) are presented in table 1.

Field Sampling

Sampling was conducted during February and March of 1993 prior to harvesting and again in 1997 approximately 3.5 years after harvesting. Five sampling locations were systematically located 11.4 m from each subplot center. Sampling locations were relocated if abnormal conditions occurred, such as large surface rocks, woody debris more than 7.6 cm in diameter, or previous manmade disturbances (e.g. old roads, etc.). Thus, samples and results reflect forest floor conditions from undisturbed areas that were not dominated by rocks or woody materials. Less than 5 percent of the sample locations had to be relocated due to these criteria.

The forest floor, excluding woody debris greater than 7.5 cm in diameter, was collected within a 0.1-m^2 square frame at each of five sampling locations. Two layers or stages of decomposition were recognized: (1) a litter L layer, which included the uppermost, relatively undecomposed material that was mostly deposited in the autumn pulse of litterfall and (2) a fermentation F layer consisting of partially decomposed, older material located between the soil surface and the L layer. The L and F layers are also frequently referred to as the O_i and O_e horizons, respectively. The color, texture, and level of fragmentation of foliage (especially the hardwoods) were used to define the boundary between the L and F layers. The boundary between the bottom of the F layer and the soil surface was also based on decomposition stage. The F layer contains fragments of vegetation that could be identified. By contrast, the soil surface was either mineral soil or dark, amorphous organic matter, representing the humus H layer or the O_a horizon. A well defined H layer rarely exists in the forest floors of southern forests because of rapid decomposition rates and incorporation of organic matter within the soil by fauna (Switzer and others 1979). Thus, any H layer material present was not included within the forest floor sample.

Laboratory Procedures and Analysis

Forest floor samples were dried to a constant weight at 75 °C and weighed. Each L layer sample was separated into woody and foliar components. The woody component (WD) included branches, bark, small stems, and reproductive material (e.g., pine cones). The foliar component of each sample was separated into pine (PF) and hardwood foliage (HF) and weighed. Thus, the L layer was represented by PF, HF, and WD in 1993. As a result of the prolific increase in herbaceous plants in the harvested areas, a miscellaneous class of L layer was added in 1997. All forest floor woody material from the clearcuts was classified as F layer due to its highly degraded form 3 years after harvesting and the lack of any current year inputs of woody material.

The forest floor material for a given component was composited for each subplot and ground to pass a 20-mesh sieve. Loss on ignition was determined by heating samples at 500 °C for 4 hours. This is a frequently used approximation of organic matter and is commonly expressed as a percent of total weight. Forest floor concentrations of P, K, Ca, Mg, Mn, S, and Fe were determined by inductance coupled plasma analysis after nitric/perchloric digestion (University of Arkansas, Soil Test Laboratory, 1990a). Total N concentrations were determined using a Tecator Kjeltec Model 1030 Auto Analyzer after sulfuric acid/hydrogen peroxide digestion (University of Arkansas, Soil Test Laboratory, 1990b).

Statistical Design and Methods

To evaluate whether different regeneration cutting methods altered forest floor characteristics, a randomized complete block analysis of variance was used to evaluate differences among study treatments. Ecoregions, as described by Baker (1994), were used as the blocking factor. Dependent variables used for the analysis were either individual post-harvest measurements or differences between pre-harvest and post-harvest measurements (1993-1997). The differences in pre- and post-harvest measurements were used to evaluate if changes in various forest floor components after harvesting were similar or differed among treatments. If differences among treatments were significant (α=0.05), Dunnet's multiple range test (Dunnet 1964) was used to compare the magnitude of changes of the harvested treatments to any changes in the uncut controls. Significant differences between a given harvesting treatment and the uncut treatment indicated changes in a forest floor component or nutrients were of greater magnitude than could be attributed to normal annual variation or inconsistencies in forest floor sampling techniques.

Table 2—Mean mass, organic matter content, and loss on ignition of the total forest floor, L layer, and F layer for each treatment in 1993 and 1997

	CC		PSW		PHSW		PSTS		PHSTS		UC	
Layer	1993	1997	1993	1997	1993	1997	1993	1997	1993	1997	1993	1997
Mass (*Mg ha*$^{-1}$)												
Total	22.4	21.1	25.1	22.6	24.9	21.8	26.2	21.4	28.2	19.5	24.0	20.5
L	4.4	1.3	5.7	2.5	5.5	2.8	6.0	1.8	5.1	2.4	5.5	2.9
F	18.0	19.8	19.5	20.1	19.3	19.0	20.1	19.7	23.1	17.1	18.5	17.6
Organic matter content (*Mg ha*$^{-1}$)												
Total	17.8	15.0	19.3	18.3	19.8	17.9	19.9	17.3	19.1	16.6	19.0	17.8
L	4.3	1.1	5.4	2.4	5.3	2.6	5.8	1.7	5.0	2.3	5.3	2.8
F	13.6	13.9	13.8	16.0	14.4	15.2	14.1	15.6	14.2	14.4	13.7	15.0
Percent loss on ignition												
Total	80.1	71.1*[a]	77.7	81.5	79.2	81.6	76.6	81.8	69.0	85.2	79.1	86.4
L	96.4	90.9*	95.5	96.0	96.0	95.0	96.2	95.6	96.2	96.1	96.0	96.8
F	76.0	69.7*	72.8	79.7	74.6	79.3	70.5	80.6	62.9	83.8	73.2	84.7

L = litter layer; F = fermentation layer; CC = clearcut; PSW = pine shelterwood; PHSW = pine-hardwood shelterwood; PSTS = pine single-tree selection; PHSTS = pine-hardwood single-tree selection; UC = unharvested control.

[a] Changes between preharvest (1993) and postharvest (1997) for a given forest floor attribute and regeneration cutting method denoted by * significantly differ ($\alpha = 0.05$) from those observed in the uncut control.

RESULTS AND DISCUSSION

Mass and Organic Matter

Mass of all forest floor components and organic matter content of the L layer were generally lower, while the loss on ignition of the total forest floor and F layer was greater in 1997 than in 1993 (table 2). These changes were relatively consistent among all treatments including the uncut stands. This variation in mass and loss on ignition appears to be related to a subtle difference in sampling methodologies used in 1993 compared to 1997. A greater portion of the forest floor was designated as the F layer and potentially less of the H layer/mineral soil interface was included with the F layer in 1997 than in 1993. Thus loss on ignition and organic matter contents were greater in the F layer, while organic matter content and mass were lower in the L layer in 1997 than in 1993.

These changes in sampling methods were caused by the manner in which the crews collected and delineated samples in 1993 and 1997. Although one original member of the 1993 crew helped to train the 1997 crew, differences in collection methods were still evident. Assuming annual fluctuations in the amount or composition of forest floor are relatively minor in uncut stands, any changes in these parameters observed in the uncut stands between 1993 and 1997 most likely represent differences in sampling methodologies. Thus, any changes in forest floor characteristics within the harvested stands greater or less than those in the uncut control should represent actual differences related to silvicultural practices and harvesting, rather than sampling methodology.

Harvesting did not appear to significantly change total forest floor, L layer, or F layer mass or organic matter content. Differences between 1993 and 1997 values in the harvested stands were similar to those observed in the uncut stands (table 2). However, clearcutting significantly reduced loss on ignition of the L layer and F layer. Due to the alteration in sampling procedures, loss on ignition in 1997 was greater or approximately equal to 1993 values in the uncut control. Differences in pre- and post-harvest loss on ignition were reduced in the shelterwood and clearcut regeneration treatments. In the case of the clearcut, 1993 values were significantly greater than the 1997 values. Harvesting often churns and mixes mineral soil with the forest floor (Mroz and others 1985; Alban and Perala 1990; Liechty and others 1992). Thus, part of this decrease in loss on ignition could be attributed to mixing of mineral soil with forest floor as a result of harvesting and associated disturbances.

The changes in loss on ignition could also be related to the changes in composition of the forest floor. Figure 1 and table 3 indicate that composition of the L layer was altered by the harvesting treatments. The amount of litter classified as miscellaneous was greatest in the treatments that had the highest amount of tree removal (CC, PSW, and PHSW). This increase in the miscellaneous L layer component occurred with the rapid occupation of harvested sites by herbaceous plants. The clearcut had the largest amount of herbaceous vegetation and the lowest amounts of PF in the L layer. This modification to litter inputs appeared to have altered the character and physical attributes of the forest floor in the stands that received the greatest amount of tree removal. Loss on ignition of miscellaneous material was lower than HF, which was lower than PF (table 3). Evidently, the reduction in the loss of ignition in the clearcut was caused by: (1) the reduction in PF, (2) the increase in

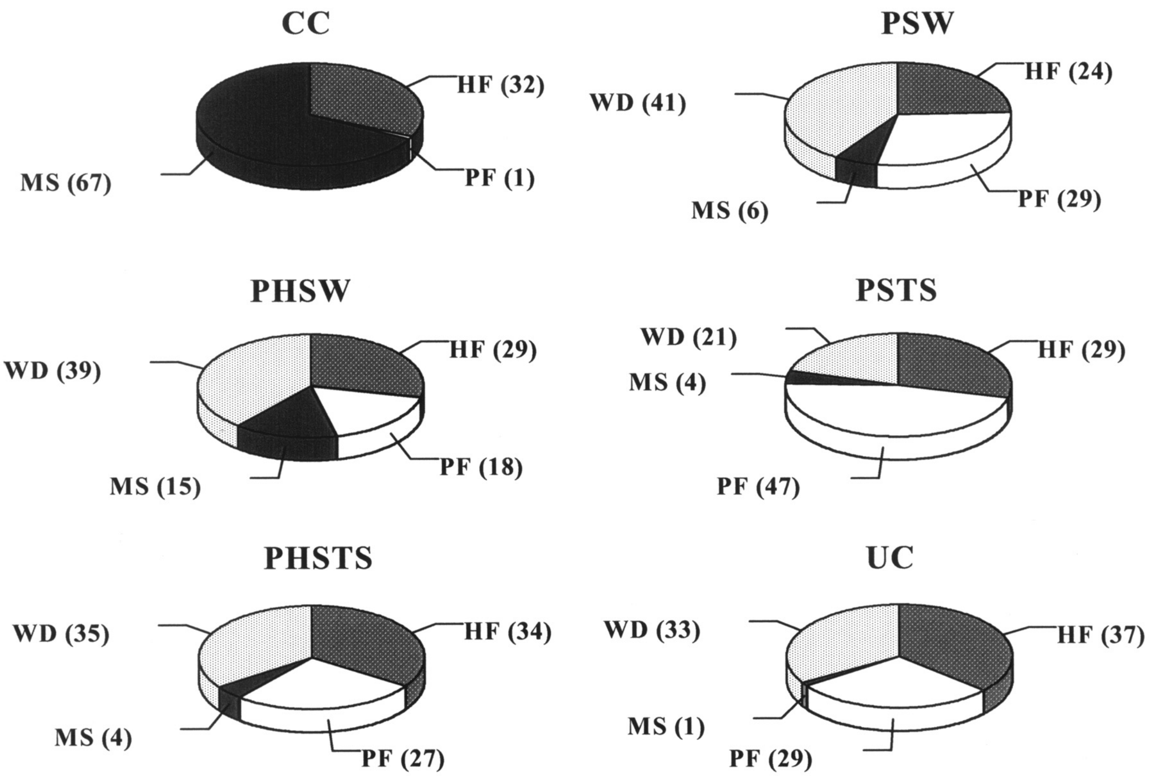

Figure 1—Proportion of mass (percent) in L layer hardwood foliage (HF), pine foliage (PF), woody (WD), and miscellaneous (MS) components by reproduction cutting method in 1997; CC = clearcut, PHSW = pine-hardwood shelterwood, PHSTS = pine-hardwood single-tree selection, PSW = pine shelterwood, PSTS = pine single-tree selection, UC = unharvested control.

Table 3—Mean mass and loss on ignition for the L layer hardwood foliage, pine foliage, and miscellaneous components in 1997

Component	CC	PSW	PHSW	PSTS	PHSTS	UC
			Mass ($Mg\ ha^{-1}$)			
HF	0.41*[a]	0.60*	0.81	0.51*	0.81	1.06
PF	0.01*	0.70	0.50*	0.82	0.63	0.83
MS	0.85*	0.15	0.42	0.07	0.10	0.04
			Percent loss on ignition			
HF	95.0	94.5	93.4	94.2	94.5	94.2
PF	96.4	96.6	96.5	96.1	96.5	96.1
MS	89.8*	94.2	92.2	93.4	94.6	94.3

CC = clearcut; PSW = pine shelterwood; PHSW = pine-hardwood shelterwood; PSTS = pine single-tree selection; PHSTS = pine-hardwood single-tree selection; UC = unharvested control; HF = hardwood foliage; PF = pine foliage; MS = miscellaneous components.

[a] Treatments for a given with an * are significantly (α = 0.05) different than the uncut control.

miscellaneous material which has the lowest loss on ignition, and (3) the mixing of mineral soil and forest floor.

The increased amount of herbaceous vegetation also appeared to reduce potential changes in forest floor mass attributed to the removal and harvesting of trees within the clearcut treatments (table 3). In the clearcut during 1997 approximately 62 percent of the L layer mass or 0.85 Mg ha^{-1} was classified as miscellaneous. Without the input of this material to the forest floor and ultimately to the soil, organic pools would likely have been significantly lower with this regeneration method.

A change in forest floor composition was also noted for the pine only treatments (PSW and PSTS). In each of these two treatments, hardwood removal significantly lowered the amount of HF in the L layer (table 3). However, these changes in composition did not appear to significantly alter loss on ignition of the forest floor or the L layer.

Nutrient Concentration

Harvesting treatments also altered the chemical composition of the forest floor. Most conspicuous was a decrease in N, P, and S concentrations of the overall forest floor and the F layer within the clearcut stands (table 4). N, P, and S concentrations in the F layer and the forest floor of the uncut controls in 1997 were higher or similar to concentrations

Table 4—Mean preharvest (1993) and postharvest (1997) total forest floor, L layer, and F layer nutrient concentrations for each regeneration cutting method

	CC		PSW		PHSW		PSTS		PHSTS		UC	
Layer	1993	1997	1993	1997	1993	1997	1993	1997	1993	1997	1993	1997
							percent					
							N					
Total	0.97	0.89*[a]	0.90	0.92	0.91	0.97	0.89	0.92	0.91	1.00	0.89	1.01
L	0.68	1.01*	0.68	0.76	0.68	0.88	0.69	0.80	0.70	0.82	0.68	0.78
F	1.04	0.87*	0.98	0.95	0.98	0.98	0.95	0.93	0.96	1.01	0.95	1.05
							P					
Total	0.059	0.049*	0.055	0.049	0.057	0.050	0.052	0.048	0.055	0.052	0.052	0.052
L	0.047	0.052	0.044	0.037	0.047	0.041	0.047	0.038	0.046	0.041	0.043	0.037
F	0.061	0.049*	0.059	0.052	0.060	0.051	0.054	0.049	0.057	0.054	0.055	0.055
							K					
Total	0.09	0.06	0.10	0.06	0.09	0.06	0.08	0.06	0.09	0.06	0.07	0.06
L	0.07	0.06	0.07	0.08	0.08	0.10	0.07	0.09	0.07	0.09	0.06	0.10
F	0.10	0.06	0.11	0.06	0.10	0.06	0.09	0.06	0.10	0.06	0.08	0.06
							Ca					
Total	0.73	0.70	0.88	0.81	0.87	0.89	0.82	0.84	0.79	0.83	0.79	0.81
L	0.85	0.74	1.13	0.75	1.08	0.96	1.08	0.74	1.00	0.82	1.00	0.89
F	0.70	0.69	0.79	0.83	0.80	0.87	0.75	0.85	0.75	0.84	0.72	0.80
							Mg					
Total	0.098	0.080	0.105	0.076*	0.103	0.083	0.093	0.070*	0.123	0.093*	0.101	0.080
L	0.112	0.097	0.118	0.080*	0.122	0.101	0.118	0.087*	0.114	0.095	0.097	0.095
F	0.095	0.079	0.101	0.075*	0.098	0.081	0.086	0.068	0.125	0.093*	0.082	0.076
							S					
Total	0.10	0.08*	0.08	0.08	0.09	0.08	0.09	0.08	0.09	0.08	0.09	0.09
L	0.05	0.09	0.08	0.07	0.08	0.07	0.08	0.07	0.08	0.07	0.07	0.07
F	0.10	0.08*	0.09	0.08	0.10	0.08	0.09	0.08	0.09	0.08	0.09	0.09
							Mn					
Total	0.10	0.05*	0.14	0.09*	0.12	0.10*	0.14	0.09*	0.12	0.08*	0.12	0.14
L	0.08	0.05	0.12	0.07*	0.11	0.08	0.12	0.08*	0.09	0.07	0.11	0.11
F	0.10	0.04*	0.15	0.10*	0.13	0.11	0.14	0.09*	0.13	0.08*	0.13	0.14
							Fe					
Total	0.39	0.45	0.50	0.33	0.37	0.32	0.33	0.24	0.53	0.21	0.38	0.15
L	0.02	0.12	0.04	0.02	0.02	0.08	0.02	0.02	0.02	0.01	0.02	0.01
F	0.48	0.48	0.63	0.37	0.47	0.38	0.42	0.26	0.65	0.24	0.05	0.17

L = litter layer; F = fermentation layer; CC = clearcut; PSW = pine shelterwood; PHSW = pine-hardwood shelterwood; PSTS = pine single-tree selection; PHSTS = pine-hardwood single-tree selection; UC = unharvested control.

[a] Changes between preharvest (1993) and postharvest (1997) forest floor concentrations for a given regeneration cutting method denoted by * significantly differ ($\alpha = 0.05$) from those observed in the uncut control.

measured in 1993. In contrast concentrations of these three nutrients in the forest floor of the clearcuts generally decreased after harvesting (table 4). Decreases in concentrations of these nutrients appeared to be strongly related to the observed decreases in loss of ignition. N, P, and S are important elements in organic constituents, such as proteins, amino acids, etc. Mixing of mineral soil, which has N, P, and S concentrations, with the forest floor would significantly reduce the concentrations of these elements in addition to lowering loss on ignition. However, mixing of mineral soil with forest floor did not significantly lower base concentrations due to the greater similarity of concentrations of bases in these two materials.

Although loss on ignition decreased after harvesting in both the L layer and the F layer, concentrations of N, P, and S in the L layer generally increased after clearcutting. Average concentration of N, P, and S in the clearcut stands were respectively 0.68, 0.047, and 0.08 percent prior to harvesting in 1993 and 1.01, 0.052, and 0.09 percent after harvesting in 1997 (table 4). Increases of N were significantly greater than those found in the uncut stands (table 4). These increases in concentration were related to the decrease of PF and the corresponding increase of HF and miscellaneous material in the L layer within the clearcuts (figure 1 and table 5). Concentrations of N, P, and S were 36 to 50 percent greater in HF and miscellaneous material than in PF (table 5). Thus while mixing of mineral soil and forest floor tended to decrease N, P, and S in the F layer, the reduction of PF increased the concentrations of these nutrients in the L layer. The reduction in N, P, and S levels in the F layer was apparently great enough to decrease the overall concentrations of these elements in the forest floor as a whole.

Regeneration treatments that retained pine and excluded hardwoods also altered the forest floor nutrient levels. Decreases in forest floor concentrations of Mg and Mn in the PSW and PSTS treatments after harvesting were significantly greater than decreases observed in the uncut stands (table 4). Tree removal at any level was found to decrease the concentrations of these nutrients in the forest floor. However, changes were always greatest in the PSW and PSTS treatments. Concentrations of bases, such as Mg and Mn, were consistently higher in hardwood compared to PF (table 5). Exclusion of HF with an accompanying dominance of PF ultimately decreased levels of Mg and Mn in the forest floor. This alteration of chemistry of the forest floor was most evident in the L layer but changes in concentrations in the F layer and the total forest floor were also evident. Changes in concentrations of Mg and Mn of the F layer in the PHSTS were also significant. Calcium levels were reduced in PSW and PSTS treatments, but reductions in concentrations after harvesting were not significantly greater than those found in the uncut stands (table 4). The prolific growth of herbaceous vegetation did not appear to have a significant impact on base concentrations of the forest floor. Average concentrations of Ca, Mg, and Mn in the miscellaneous L layer component were similar to concentration of these elements in PF (table 5).

Reductions in forest floor bases, such as Mg and Mn, could not solely be attributed to the increased dominance of PF in the forest floor. Concentrations of these elements in HF and PF were also reduced after harvesting (table 6). A reduction of Mg in HF was evident within all harvesting treatments. Declining levels of Mg in PF were significantly greater than reductions observed in the uncut controls within the PSW and PSTS treatments only. Declining levels of Mn were also evident in HF and PF collected from the harvested treatments. Reductions were only significantly greater than differences in uncut control for the clearcut and PSTS treatments.

The diminished levels of Mg and Mn in the hardwood and pine component of the L layer may potentially be attributed to two specific changes related to harvesting. First, harvesting can result in accelerated soil leaching of NO_3^- (Vitousek 1981; Van Lear and others 1990) and cations (Kimmins 1996). Accelerated leaching of Mg and Mn may have reduced the availability of these two cations in soil and thus uptake of these nutrients by the residual trees. Reduction in uptake can lower levels of nutrients in foliage when original available nutrients in soil are at or below levels required by the residual stand. Harvesting can also change the composition of HF in the forest floor. For example, removal of undesirable species in favor of more desirable species in the PHSW and PHSTS treatments could have altered HF composition. Treatments that retain minimal residual densities dramatically increased the light environment within the stands. These conditions favored growth and establishment of early successional, shade intolerant hardwoods as well as shrubs and other herbaceous vegetation. Early successional hardwood species that respond to the disturbances often have foliage physiology and nutrient levels much different than later successional, shade tolerant species (Hinesley and others 1991; Kozlowski and others 1991).

Table 5—Mean nutrient concentration in L layer hardwood foliage, pine foliage, and miscellaneous components from all treatments in 1997

Component	N	P	K	Ca	Mg	S	Mn	Fe
	- - - - - - - -	- - - - - - - -	- - - - - - - -	- - - -	*percent* - - - -	- - - - - - - -	- - - - - - - -	- - - - - - - -
HF	1.06	0.049	0.07	1.16	0.13	0.09	0.12	0.03
PF	0.75	0.042	0.09	0.46	0.09	0.06	0.08	0.01
MS	1.12	0.057	0.09	0.67	0.08	0.09	0.06	0.09

HF = hardwood foliage; PF = pine foliage; MS = miscellaneous components.

Table 6—Concentrations of Mg and Mn in L layer hardwood and pine foliage components

	CC		PSW		PHSW		PSTS		PHSTS		UC	
Component	1993	1997	1993	1997	1993	1997	1993	1997	1993	1997	1993	1997
	percent											
	Mg											
HF	0.16	0.11*[a]	0.18	0.14*[a]	0.14	0.09*[a]	0.17	0.12*[a]	0.18	0.14*[a]	0.15	0.14
PF	—[b]	—[b]	0.12	0.08*[a]	0.12	0.09	0.12	0.08*[a]	0.12	0.10	0.10	0.09
	Mn											
HF	0.14	0.06*[a]	0.19	0.12	0.19	0.12	0.20	0.11*[a]	0.17	0.11	0.21	0.18
PF	—[b]	—[b]	0.10	0.08	0.09	0.07	0.09	0.07	0.08	0.07	0.10	0.10

L = litter layer; HF = hardwood foliage; PF = pine foliage.

[a] Absolute differences between concentrations for pre- and post-harvest sampling (1993 and 1997) denoted with * are significantly greater than differences for uncut control ($\alpha = 0.05$).

[b] There was not sufficient pine foliage for chemical analysis in the clearcut stands.

The study design did not let us directly determine which of these factors were responsible for the changes in foliar chemistry. Since reductions in PF Mg also occurred after harvesting and shortleaf pine was the dominate species prior to and after harvesting, it seems likely that changes in concentrations of Mg were at least in part due to changes in availability of Mg in the soil. However, greater reductions in concentrations occurred in HF and a greater number of treatments showed significant reductions in concentrations of Mg and Mn in HF than PF. These responses would also support the hypothesis that a change in hardwood species composition has altered HF chemistry within the L layer. It is likely that both nutrient availability and species composition has changed in the harvested stands during the study.

Reductions in nutrient concentrations other than Mg were not significant. N concentrations of PF in the PSW treatment and HF concentrations of Fe in the clearcut stands significantly increased after harvesting. Increases in concentrations of these elements within the other treatments were not significantly greater than in the uncut stands. Thus increases in N and Fe were considered anomalies.

Nutrient Content

Similar to mass, nutrient contents of the forest floor were generally lower in all treatments in 1997 compared to 1993 (table 7). Decreased nutrient contents were related to the differences in sampling methodology and potentially natural changes in annual fluxes of litter. The lack of any significant reductions in nutrient content within the forest floor appears to be related to a high variation in forest floor mass among plots, treatments, and sampling periods rather than the lack of a significant change in nutrient concentrations (table 4). Significant reductions of Mg and Mn in the F layer of the PHSTS treatment and Mn in the F layer of the PSW were evident. However, substantial decreases in mass of the F layer along with changes in concentrations were responsible for changes within the PHSTS treatment. Reductions in the Mn contents of the PSW treatment reflect changes in concentrations rather than mass.

Reductions in contents of N, P, Ca, Mg, S, and Mn between pre- and post-harvested periods were consistently greater in the harvested than those in the uncut stands. Even if results of the PHSTS treatment are disregarded, reductions in amounts of these elements were as much as 3-7 times greater in the harvested stands than in the uncut control. Elements which showed the greatest reductions in concentrations, such as Mn, usually had the greatest reductions in contents (tables 4 and 7). Changes in contents of the L and F layers were variable and for the most part decreased. However, Ca content of the F layer and the Fe content of the L layer in one or more of the harvested treatments increased. It seems likely from our results that nutrient contents in the forest floor decreased after harvesting but these decreases were not substantial enough to be statistically significant.

Given the high variability within forest floor mass and contents, we cannot accurately estimate the amount of loss associated with a given harvesting treatment. To better determine the significance of these losses, differences in contents between pre- and post-harvest periods were adjusted for observed differences in the uncut control by subtracting average uncut control differences. These adjusted values were then expressed as a percent of the original 1993 content for a given treatment. The PHSTS treatment had the greatest reduction in forest floor mass (18.5 percent) and generally the greatest reductions in nutrient content. Reductions in Mn were substantial for all harvesting treatments. If the PSTS treatment is disregarded, reductions in other nutrients were generally less than 20 percent of original pre-harvest contents. Mg and K also consistently showed relatively high reductions in all or the majority of treatments. There appeared to be no specific trends in nutrient content reductions related to intensity of tree removal as indicated by target residual densities. The clearcut stands that had the greatest amount of tree removal often showed the lowest nutrient content reduction.

Table 7—Preharvest (1993) and postharvest (1997) total forest floor, L layer, and F layer nutrient contents for each regeneration cutting method

	CC		PSW		PHSW		PSTS		PHSTS		UC	
Layer	1993	1997	1993	1997	1993	1997	1993	1997	1993	1997	1993	1997
	kg ha^{-1}											
							N					
Total	216	187	223	206	227	211	230	195	254	197	213	206
L	30	13	38	18	37	25	41	14	36	19	38	22
F	186	174	185	188	190	186	190	181	219	178	175	183
							P					
Total	13.1	10.3	13.7	10.8	14.1	10.8	13.3	10.2	15.2	10.2	12.5	10.7
L	2.1	0.7	2.5	0.8	2.6	1.1	2.6	0.7	2.3	1.0	2.5	1.0
F	10.9	9.7	11.2	10.0	11.6	9.6	10.7	9.5	12.9	9.3	10.0	9.6
							K					
Total	20.5	12.0	26.3	13.3	22.8	14.2	21.0	13.3	26.4	11.9	17.4	12.7
L	3.2	0.8	4.1	1.4	4.3	2.3	4.1	1.4	3.7	1.8	3.3	1.7
F	17.3	11.2	22.2	11.9	18.4	12.0	16.9	11.9	22.7	10.1	14.1	11.0
							Ca					
Total	167	148	215	182	212	196	206	174	221	165	187	162
L	39	9	63	19	62	28	60	13	54	20	58	26
F	129	139	152	163	150	168	146	161	168	145	130	136
							Mg					
Total	22.3	17.2	26.5	16.6	25.5	17.7	23.7	14.7	34.0	17.6*^{a}	20.4	15.7
L	5.0	1.1	6.5	1.9	6.7	2.9	6.7	1.5	6.0	2.3	6.0	2.8
F	17.2	16.1	20.0	14.8	18.8	14.7	17.0	13.2	28.1	15.3*	14.8	13.0
							S					
Total	21.8	16.7	21.1	17.3	23.4	16.9	22.7	16.6	25.4	16.3	21.7	18.2
L	3.5	1.2	4.5	1.5	4.4	2.0	4.5	1.2	4.0	1.7	4.2	2.1
F	18.3	15.5	16.6	15.8	19.0	15.0	18.2	15.4	21.4	14.7	17.5	16.1
							Mn					
Total	22.6	9.7	36.5	20.4	30.0	21.7	33.1	19.1	33.1	16.1	28.6	27.4
L	3.9	0.7	6.4	1.6	5.8	2.1	6.6	1.3	4.8	1.7	6.0	3.1
F	18.7	9.0	30.2	18.8*	23.8	19.6	26.4	17.7	28.3	14.5*	22.5	24.3
							Fe					
Total	88	96	134	76	91	68	84	61	152	38*	90	29
L	1	2*	2	1	1	1	2	0	1	0	1	0
F	887	94	132	75	90	68	82	60	151	38	88	29

L = litter layer; F = fermentation layer; CC = clearcut; PSW = pine shelterwood; PHSW = pine-hardwood shelterwood; PSTS = pine single-tree selection; PHSTS = pine-hardwood single-tree selection; UC = unharvested control.

[a] Changes between preharvest (1993) and postharvest (1997) forest floor concentrations for a given regeneration cutting method denoted by * significantly differ ($\alpha = 0.05$) from those observed in the uncut control.

CONCLUSION

Several of the reproduction/cutting methods evaluated in this study were found to alter the characteristics and nutrient compositions of the forest floor. Clearcutting generally was found to increase mixing of forest floor and mineral soil as well as the proliferation of herbaceous vegetation. This resulted in decreasing concentrations of N, P, and S in the forest floor. Regeneration methods that removed the majority of hardwoods tended to decrease levels of Mg and Mn in the forest floor. These decreases were attributed to the increased proportion of PF with lower concentrations of Mg and Mn compared to HF. However, reductions in concentrations of these nutrients were also evident in HF of all of the harvested stands. Reductions in Mn content of the forest floor in the harvested stands were between 22-52 percent but reductions in content of other nutrients were generally less than 20 percent.

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Session 5

Phase II Understory Vegetation and Visual Quality Research

Dan Nolan, Moderator

SCENIC BEAUTY IN SUMMER THE YEAR BEFORE, THE YEAR AFTER, AND 4 YEARS AFTER HARVEST

Victor A. Rudis, Hochan Jang, and James H. Gramann[1]

Abstract—This study examined the effects of four silvicultural options (clearcut, group selection, shelterwood, and untreated) on the perceived scenic beauty of shortleaf pine-oak (*Pinus echinata-Quercus* spp.) stands of national forest land in the Ouachita Mountains of Arkansas. Twelve randomly selected stands were selected and nine were cut in the winter of 1993. Color images of the stands were captured in June 1992, July 1994, and July 1997. Texas A&M University undergraduate students rated the scenic beauty of five images per plot and summer season results are reported here. The visual impact of cutting these stands was severe a year (two growing seasons) after treatment, with intensive treatments yielding the lowest scenic beauty preferences. Four years after treatment, however, no statistical differences were detected among treated and untreated stands. Results are preliminary, but clearly show that the negative visual aspects of harvesting declines with time.

INTRODUCTION

An attractive, healthy, and visually pleasing natural environment is closely related to the success of tourism and recreation-related industries (Mieczkowski 1995). Such environments improve the quality of life for nearby communities and raise their property values (Correll and others 1978, Crompton 1993, Ulrich and others 1991a, 1991b).

Ecosystem management, a paradigm adopted by the U.S. Department of Agriculture, Forest Service, National Forest System, emphasizes the integration of social, economic, and ecological needs at different scales of time and space (Salwasser and Pfister 1994). Society places importance on the public enjoyment of forests, especially their scenic values (Brunson 1991). Because people's first response to the forest is often visual, and their evaluation of place is also visual (Gobster 1993), they judge forest management and policy on a forest's appearance (Hull 1989).

In managing forests, knowledge of the visual impact and speed of recovery from alternative harvest disturbances are critical elements of an appearance-sensitive approach to ecosystem management. Empirical studies have confirmed the validity and reliability of photographs to represent scenes of field conditions (Brown and others 1989, Buhyoff and Wellman 1979, Hoffman and Palmer 1995).

To develop hypotheses about a variety of disturbances, Benson and Ullrich (1981) studied preferences of foreground, or near-stand views of Douglas fir-larch and lodgepole pine forest types in Montana and Wyoming having a range of stand histories and treatments. Viewers rated scenic values lower for stands with severe cutting and burning activities, and rated scenic values higher in stands where time-since-disturbance was 25 to 50 years old. Had Benson and Ullrich (1981) conducted long-term monitoring and controlled experiments at the same location—or in forest types of the Eastern United States—results might have been different.

Rudis and others (1994, 1999) monitored 0.5 acres of experiment-scale (1.6 acres) treatments in shortleaf pine-oak vegetation types of the Ouachita Mountains of Arkansas and found measurable differences in vegetation structure with the type and amount of cutting two growing seasons after harvest. Gramann and Rudis (1994) found the amount of cutting in these same areas negatively associated with scenic values of within-stand views. In operational-scale (about 40 acres) treatments elsewhere in the Ouachita Mountains, the amount of cutting was negatively associated with scenic values within a year of cutting (Barlow and Rudis, in press) and a full year (two growing seasons) after cutting (Li and others, in press), but longer term trends had not been quantified.

OBJECTIVES AND THE STUDY AREA

We monitored change in scenic beauty the summer before, the year (two growing seasons) after, and 4 years after cutting disturbance and compared the effects of different harvest practices. Study locations were part of a 52-stand study region (Baker 1994, Guldin and others 1994, Mersmann and others 1994) in national forests in the Ouachita Mountains of Arkansas. Treated stands were harvested in the winter of 1992-93 and subsequent reproduction treatments occurred in 1994. Silvicultural treatments were randomly assigned to 1 of 13 stands in each quadrant of the study region (Baker 1994). Cost and time constraints limited our study to four treatment conditions and the north, east, and south quadrants of the study region. The conditions and stand references for this study were

(a) Untreated: untreated stands (CON) retained in their natural state, averaging 129 square feet per acre. The north, east, and south stands were 0284-11, 0605-05, 0023-10, respectively

(b) Group selection: pine-hardwood group selection (PHGS): 60 square feet per acre retained (70 square feet per acre removed); openings 0.1 to 2.0 acres.

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Retention of 20 to 45 trees per acre shortleaf pine and 10 to 20 trees per acre hardwoods. The north, east, and south stands were 0046-18, 1124-11, 0035-42, respectively

(c) Shelterwood: pine-hardwood shelterwood (PHSW): 40 square feet per acre retained (90 square feet per acre removed). Retention of 10 to 30 trees per acre shortleaf pines and 10 to 30 trees per acre hardwoods. The north, east, and south stands were 0457-12, 1119-21, 0027-01, respectively

(d) Clearcut: all merchantable trees removed (CC): 0 to 5 square feet per acre of trees retained (129 square feet per acre removed). The north, east, and south stand compartments were 0458-16, 1067-15, 1658-05, respectively.

METHODS

We took about 15 images per stand within a 2-week period. Viewing points were at five to six predetermined locations (depending on the size of the stand) and spaced evenly within each stand (Baker 1994). Equipment used included ASA 400, 35-mm transparency film, push processed to ASA 800, and an f2.8-lens (Olympus XA) camera. Duplicate, overexposed or underexposed images were discarded. This study used about five randomly selected scenes per stand from the remaining images.

We tested scenic beauty at three time periods for four conditions: one untreated (control) and three treatment conditions; and three replicates per condition: one in each of the north, east, and south quadrants of the study region. The season and year for images included a pretreatment summer sample, a year (two growing seasons) after treatment in spring, summer, fall, and winter, and 4 years after treatment in summer, fall, and spring. For brevity, analysis reported here used data only from the summer season; quadrant and season differences are not included.

Rating sessions followed procedures developed by Daniel and Boster (1976) and used rating-analysis software (Brown and others 1990). In each session, raters viewed 80 images, 20 of which were baseline images shown to every group. These baselines were placed in every fourth position of the slide carousel. Raters were shown each scene for 8 seconds, then asked to rate each image on a scale of 1 to 10, where 10 was the highest scenic beauty. The score for an image represented the average scenic beauty rating, called SBE, relative to baseline images shown to all raters (Brown and Daniel 1990). People's perception, represented by SBE, is widely used in visual impact estimation (Brown and Daniel 1990, Ribe 1989). However, one is often cautioned that no true interval exists. Commonly, the resulting ordinal scores are assumed to provide adequate precision and not violate assumptions of standard statistical tests.

We conducted two sets of three rating sessions. Each session consisted of about 5 warm-up images, 20 baseline images, and 80 images stratified by stand and by 4 points-in-time. The first set was from summer 1994, fall 1994, winter 1995, and spring 1995. The second set was from summer 1992, summer 1997, fall 1997, and spring 1998. The total set was 480 images. The baseline images were from four seasons in 1994 and 1995 from another national forest region in the Ouachita Mountains (Winona Ranger District); Gritter (1997) also used these images. Thirty-one students at Texas A&M University viewed a portion of the images and asked to rate scenes as part of a classroom assignment. Respondents were students taking undergraduate courses in parks and diverse populations; methods of park, tourism, and recreation research; and management of tourism and recreation enterprises. Students were predominantly from the Southeastern United States, and majored in agricultural development, architecture, or recreation, park, and tourism sciences.

We used standardized SBE scores, called SBEz, to assure a uniform scale among different raters (Brown and Daniel 1990) and used nonbaseline image SBEz values for subsequent calculations. For ease of interpretation, we converted ratings to rankings for conducting an analysis of variance by year and treatment, calculation of averages, and F- and t-tests of significance at the 0.05 probability level. Statistical software employed SAS's General Linear Model (SAS Institute, Inc. 1990). By year, significant tests among treatments by year used the Tukey-Kramer option to account for multiple comparisons (SAS Institute, Inc. 1990).

RESULTS AND CONCLUSIONS

A visual inspection of color images suggests that overt evidence of disturbance was negatively associated with scenic beauty estimates. (A subset of images in various seasons and related SBEz values are included in the appendix.) The amount of sky in the scene changed dramatically in treated stands, with distant views apparent in areas that were extensively cut. Evidence of cutting disturbance was apparent in the year following treatment, but evidence was not readily visible after 4 years.

Quantitatively, our results showed significant differences in scenic beauty preferences among forest management activities that varied by year (table 1). The chief difference was

Table 1—Analysis of variance in summer scenic beauty ranking by year and treatment, Ouachita-Ozark National Forests

Source	Degrees of freedom	Mean square variance	F value	P (larger F)
Treatment	3	4,428.49	2.08	0.105
Year	2	5,067.44	2.38	0.096
Year by treatment	6	16,430.42	7.71	< 0.001
Residual	167	2,131.29		

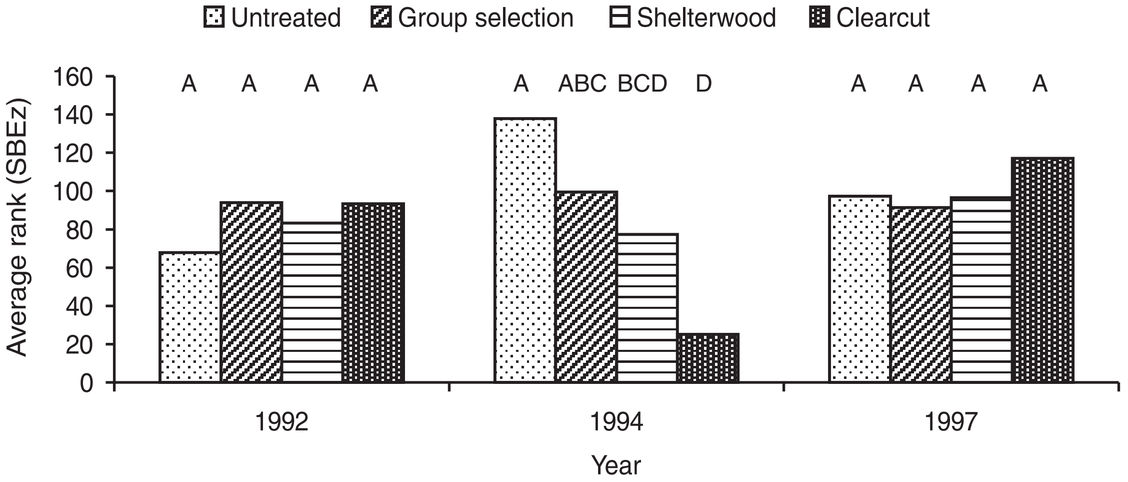

Figure 1—Average rank (standardized scenic beauty estimates) in summer scenic beauty the year before (1992), the year after (1994), and 4 years after harvest (1997) by type of harvest, Ouachita-Ozark National Forest. By year, averages among harvest treatment with the same letter are not significantly different.

in the year after treatment, with more intensive treatments yielding lower scenic beauty (fig. 1). Before treatment and 4 years after treatment, we did not detect significant differences. Failure to detect significant differences in summer between untreated stands and group selection suggests this treatment may be the least offensive of the other treatments. A year after treatment, differences in scenic beauty between group selection and shelterwood were not significant. Analyses that account for quadrant differences and preference ratings from additional images taken in other seasons may reveal other fine-scaled differences. Nevertheless, results to date corroborate a common understanding that, with time, the negative visual aspects of harvests are indistinguishable from untreated stands.

Benson and Ullrich's (1981) hypothesis for Douglas fir-larch and lodgepole pine forests was that the vegetation change in the years following treatment reduces the negative aspects of harvest disturbance. Our results for the summer season in shortleaf pine-oak forest types in Arkansas show definitively that scenic beauty is indistinguishable from untreated stands 4 years after harvest. This contrasts with data in Benson and Ullrich's (1981) report on scenic preferences, which suggested recoveries of lodgepole pine at 3 to 10 years, and Douglas fir-larch at 23 to 30 years at one site and 6 to 18 years at three other sites. Apart from methodological differences between the two studies, our study's shorter recovery time may be due to the more humid and longer growing period for vegetative growth in Arkansas compared with Montana and Wyoming.

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APPENDIX

The following is a subset of the study's digitally-archived images with identifying codes by treatment, stand, and vantage point. References under each image are the season and time period: 1992 = year before treatment; 1994, 1995 = year after treatment; 1997, 1998 = 4 years after treatment; and standardized scenic beauty estimate (SBEz). SBEz scores ranged from -170 to +270 for views from all seasons. For summer views, we also list the ranking of SBEz, where 1 = lowest and 179 = highest. (A seven-digit compact disk/image code is included for archival purposes.)

Untreated (Control), Stand 0284-11, Point 3

Summer (1992) SBEz = 0.06
Rank = 88 CD 2608-069

Winter (1995) SBEz = -62.88
CD 1634-084

Summer (1994) SBEz = 87.87
Rank = 143 CD 0015-022

Fall (1997) SBEz = 68.56
CD 4233-076

Summer (1997) SBEz = -24.89
Rank = 67 CD 4232-014

Spring (1998) SBEz = -16.07
CD 3171-084

Group Selection, Stand 1124-11, Point 4

Summer (1992) SBEz = 77.68
Rank = 135 CD 2609-069

Fall (1994) SBEz = -51.68
CD 1633-048

Summer (1994) SBEz = 38.12
Rank = 116 CD 2607-018

Winter (1995) SBEz = -116.77
CD 1636-072

Summer (1997) SBEz = -292.97
Rank = 2 CD 4232-031

Fall (1997) SBEz = 11.16
CD 4234-044

Shelterwood, Stand 0027-01

Point 5

Point 2

Summer (1992) SBEz = -104.99
Rank = 28 CD 2608-010

Winter (1995) SBEz = -189.09
CD 1634-065

Summer (1994) SBEz = 4.11
Rank = 91 CD 2607-058

Spring (1995) SBEz = 31.63
CD 1613-084

Summer (1997) SBEz = -68.15
Rank = 45 CD 4231-045

Spring (1998) SBEz = 75.22
CD 4235-91

Clearcut, Stand 0458-16, Point 6

Summer (1992) SBEz = 26.28
Rank = 104 CD 2608-054

Fall (1994) SBEz = -188.28
CD 1561-002

Summer (1994) SBEz = -155.81
Rank = 13 CD 0015-012

Spring (1995) SBEz = 235.67
CD 1613-046

Summer (1997) SBEz = 92.82
Rank = 147 CD 4231-066

Spring (1998) SBEz = 121.39
CD 3171-096

IMPACT OF RECENT TIMBER HARVESTS ON AUTUMN SCENIC BEAUTY OF NEAR-STAND VIEWS

Rebecca J. Ray Barlow and Victor A. Rudis[1]

Abstract—This study estimated the impact of 10 recent timber cutting regimes on the autumn scenic beauty of shortleaf pine-hardwood forests in the Ouachita Mountains of Arkansas. Scenes were photographed near forest stand edges—views typically observed by sightseeing visitors—from 36 treated areas cut the previous winter and 3 comparable untreated areas. Cutting regimes varied in hardwood retention, spatial arrangement, and harvest intensity. We averaged scenic beauty ratings from several groups of judges. Results showed that scenic beauty of autumn, near-stand views were significantly ($P<0.05$) lower and in inverse proportion to the amount of wood recently removed. Pine vs. pine-hardwood retention and differences among groups of judges had no significant effect on scenic beauty ratings among various treatments. The treatments—grouped into similar impact categories and in order of increasing negative impact—were: (1) low-impact and pine single-tree selection, (2) pine-hardwood single-tree selection, pine and pine-hardwood group selection, and pine and pine-hardwood shelterwood, (3) pine and pine-hardwood seed tree, and (4) clearcut harvest.

INTRODUCTION

Public interest in aesthetics for forest management appears in Federal legislation such as the National Environmental Policy Act of 1969, the Forest and Rangeland Renewable Resources Act of 1974, and the National Environmental Policy Act of 1976. The visual impact of cutting activities on forests is well known, but predicting the mitigating effects of different silvicultural treatments is uncertain.

Scenic beauty, as used in this study, is a measure of the aesthetic significance given to a scene by an observer. Scenic beauty is influenced by both the observer's culture and by the properties of the scene being observed (Smardon and others 1986). Reaction of observers to the aesthetics of forest scenery depends partly on the observer's ability—or lack of ability—to perceive and distinguish among different management activities (Magill 1990).

Daniel and Boster (1976) contend that scenic beauty is not totally "in the eye of the beholder," but inferred from the observer's perception of the landscape. The Scenic Beauty Estimation (SBE) method developed by Daniel and Boster (1976) is a procedure for rating the visual quality of scenes. Individuals typically rate representations of these scenes by numerically scoring their preference for the "scenic beauty" depicted in photographic images shown to them.

OBJECTIVES AND STUDY AREA

The main objectives of the study were to estimate the impact of silvicultural treatments on the scenic beauty of shortleaf pine-hardwood forest stands in the Ouachita and Ozark National Forests and to compare scenic beauty trade-offs when selecting silvicultural treatments (Mersmann and others 1994).

The study locations were part of a 9,600 square mile study area established by the USDA Forest Service on the Ouachita and Ozark National Forest land located in northwest Arkansas and eastern Oklahoma (Baker 1994, Guldin and others 1994). Quadrants of the study area corresponded to regions with similar land surface forms, potential natural vegetation, and geology. These quadrants (and nearby Arkansas cities) were: north (Danville), east (Cedar Creek), south (Mount Ida), and west (Black Fork). The USDA Forest Service randomly assigned silvicultural treatments to one of 13 stands in each quadrant of the study region, with trees mostly cut to promote pine reproduction (Baker 1994). We used the north, east and south quadrants of the region; costs and time constraints excluded consideration of the west quadrant.

Harvesting of treated stands occurred in the winter of 1992 to 1993; reproduction treatments were planned for 1994. The conditions, treatments, and their abbreviation are listed first by increasing intensity of harvesting (untreated, single-tree selection, group selection, shelterwood, seed tree, clearcut), second by an estimate of the square feet of basal area removed, and third by square feet of hardwood basal area removed:

1. CON: untreated stands retained in their natural state, averaging 130 ft^2 per ac
2. LIST: low impact single-tree selection: about 70 ft^2 per ac retained (about 60 ft^2 per ac removed). Retention of 30 to 55 trees per ac shortleaf pines and 10 ft^2 per ac hardwoods
3. PSTS: pine single-tree selection: 63 ft^2 per ac retained (67 ft^2 per ac removed). Retention of 30 to 55 trees per ac shortleaf pines and 0 to 5 ft^2 per ac hardwoods
4. PHSTS: pine-hardwood single-tree selection: 60 ft^2 per ac retained (70 ft^2 per ac removed). Retention of 20 to 45 trees per ac shortleaf pines and 10 ft^2 per ac hardwoods. Planned for 1994, vegetative management treatments, e.g., mechanical versus chemical treatment

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[1] Forester, Weyerhauser Corporation, Columbus, MS; Research Forester, USDA Forest Service, Southern Research Station, Starkville, MS 39759, respectively.

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of residual trees, were to be split (s) in PHSTSs, but not in nonsplit (ns) PHSTSns stands

5. PHGS: pine-hardwood group selection: 60 ft^2 per ac retained (70 ft^2 per ac removed); clearcut 0.1 to 2.0 ac. Retention of 20 to 45 trees per ac shortleaf pine and 10 ft^2 per ac hardwoods
6. PGS: pine group selection: 60 ft^2 per ac retained (70 ft^2 per ac removed); clearcut 0.1 to 2.0 ac. Retention of 30 to 55 trees per ac shortleaf pine and 0 to 5 ft^2 per ac hardwoods
7. PSW: pine shelterwood: 43 ft^2 per ac retained (87 ft^2 per ac removed). Retention of 20 to 40 trees per ac shortleaf pines and 0 to 5ft^2 per ac hardwoods
8. PHSW: pine-hardwood shelterwood: 40 ft^2 per ac retained (90 ft^2 per ac removed). Retention of 10 to 30 trees per ac shortleaf pines and 10 ft^2 per ac hardwoods. Planned for 1994, vegetative management treatments were to be split (s) in PHSWs, but not in nonsplit (ns) PHSWns stands
9. PST: pine seed tree: 23 ft^2 per ac retained (107 ft^2 per ac removed). Retention of 15 to 20 trees per ac shortleaf pines and 0 to 5 ft^2 per ac hardwoods
10. PHST: pine-hardwood seed tree: 20 ft^2 per ac retained (110 ft^2 per ac removed). Retention of 5 to 10 trees per ac shortleaf pines and 10 ft^2 per ac hardwoods
11. CC: clearcut: 0 to 5 ft^2 per ac of trees retained (127 ft^2 per ac removed). All trees were removed. Site preparation for planting of genetically improved shortleaf pine was scheduled to begin in 1994.

METHODS

We examined one untreated stand and stands demonstrating the 10 different timber-harvest regimes listed above. Table 1 lists stand and compartment numbers, and the square feet of basal area retained (and removed) for the 39 stands examined in this study. As outlined elsewhere (Guldin and others 1994), retained basal area estimates were expected values following treatment across the study region. We estimated removed basal area by subtraction from the average basal area prior to harvest treatment (130 ft^2 per ac) across the entire study region. Owing to planned 1994 reproduction treatments, PHSTS and PHSW harvest treatments were replicated twice.

The senior author selected three to four representative vantage points per stand from topographic sheets and located them during July 1993 field visits. These points were just outside, or up to 100 feet outside, the stand boundary. Picture taking was conducted from the selected vantage points during three weekends in October 1993. At that time, treated areas were recently disturbed, i.e., stand disturbance had occurred within a year after the cut, but without site preparation.

Stand views were photographed with ASA 100 35 mm transparency film using a single lens reflex camera mounted on a tripod with a 30 to 70 mm zoom lens fixed at 55 mm. To randomize view angles and avoid duplicating scenes, images at each location were taken at three compass directions (azimuths) toward the stand. The initial image was taken in a random azimuth toward the stand. The second and third images were taken 40 degrees to the left and right of the

Table 1—National forest stands sampled in October 1993, approximate square feet of basal area per acre retained (removed), treatment, and treatment abbreviation, Ouachita-Ozark National Forests[a]

Stands sampled by quadrant and stand—compartment number					
North	East	South	Retained (removed)	Silvicultural treatment	Treatment abbreviation
			ft^2/ac		
0284-11	0605-05	0023-10	130 (0)	Untreated (control)	CON
0367-04	1077-19	0062-08	70 (60)	(s) low-impact single-tree selection	LIST
1125-05	1044-03	1658-16	63 (67)	(s) pine single-tree selection	PSTS
0428-02	1073-10	1654-16	60 (70)	(s) pine-hardwood single-tree selection	
0070-10	0609-09	1649-13		(ns) pine-hardwood single-tree selection	PHSTS
0046-18	1124-11	0035-42	60 (70)	(ns) pine-hardwood group selection	PHGS
0014-18	1106-09	1648-01	60 (70)	(ns) pine group selection	PGS
0443-03	1097-06	0035-41	43 (87)	(s) pine shelterwood	PSW
0456-09	1094-04	1660-06	40 (90)	(s) pine-hardwood shelterwood	
0457-12	1119-21	0027-01		(ns) pine-hardwood shelterwood	PHSW
0458-10	1084-07	1646-08	23 (107)	(s) pine seed tree	PST
1036-17	1119-5S	1651-06	20 (110)	(s) pine-hardwood seed tree	PHST
0458-16	1067-15	1658-05	3 (127)	(ns) clearcut	CC

(s) = Split stands; (ns) = nonsplit stands; CON = untreated stands; LIST = low-impact single-tree selection; PSTS = pine single-tree selection; PHSTS = pine-hardwood single-tree selection; PHGS = pine-hardwood group selection; PGS = pine group selection; PSW = pine shelterwood; PHSW = pine-hardwood shelterwood; PST = pine seed tree; PHST = pine-hardwood seed tree; CC = clearcut.

[a] Reproduction treatments, planned for 1994, were to differ for half of the stand area of split (s) stands, but not nonsplit (ns) stands (Baker 1994).

initial azimuth. Acquisition of images incorporated a long depth of view by using an f 22 aperture setting, and bracketed ± 1 f-stop.

After the film was processed, duplicate, over- and underexposed images were discarded. Only "acceptable" images from which random samples were selected. To simplify analysis for this report, we ignored the initial blocking of the overall study's sample design, i.e., assignment of stands by quadrant. Twelve or more images were chosen to represent each treatment. Random sample selection resulted in one to six images representing a single stand, with most stands represented by three images.

We used groups of students at Mississippi State University to view and rate these images in the spring of 1994. Of the 196 students asked to rate a portion of the images during their class period, 96 percent turned in completed responses. Respondents were predominantly from the southeastern United States, 90 percent were male, with an average age of 21. Previous research has shown that students' visual preferences for natural scenes are representative of the general public (Schroeder and Daniel 1981).

There were three groups of student judges based on attendance in particular classes:

1. informed forestry class: 82 students (78 completed responses) enrolled in an "Introduction to Forest Survey" course. This group received instructions about the different types of harvest practices they were to see
2. senior forestry class: 64 students (63 completed responses) enrolled in a senior-level "Forest Management" course. This group received no message about treatments
3. non-forestry class: 50 students (47 completed responses) enrolled in a "Landscape Architecture Appreciation" course, primarily for non-landscape architecture majors. This group also received no message about treatments. Out of this group, more than 40 percent majored in professional golf management, 22 percent in business, 14 percent in landscape architecture, and the remaining 24 percent in other fields of study.

For the rating sessions, we followed procedures developed by Daniel and Boster (1976) and used RMRATE software (Brown and others 1990). Judges were shown each image for 8 seconds, then asked to rate the image on a scale of 0 to 9, where nine was the highest scenic beauty. In each session, judges viewed 80 different images, 20 of which were "baseline" images shown to every group. These "baselines" were placed in every fourth position of the slide carousel. The score for an image represented the average scenic beauty rating, called SBE, relative to "baseline" images shown to all judges (Brown and Daniel 1990). Subsequent analysis then used ratings only of nonbaseline images.

We used standardized SBE scores, called SBEz, to assure a uniform scale among different groups of judges (Brown and Daniel 1990). A widely accepted and commonly used practice in social science preference studies is to assume resulting ordinal scale preference scores are interval data, with an implied uniform distance between two adjacent scores (Daniel and Vining 1983). Furthermore, ratings with from 20 or more individuals, when normally distributed, have been shown to provide adequate precision and do not seriously violate assumptions of standard statistical tests and procedures (Daniel and Vining 1983). However, one is often cautioned that no true interval exists. That is to say, a difference in rating of 1.0 between 10.0 and 11.0 does not necessarily represent the same difference between 90.0 and 91.0.

Traditional parametric tests—where random samples are taken from one or more populations—assume an underlying normal distribution, but ratings may not follow such a distribution. The distribution of ratings was tested for kurtosis (0=not skewed and normally distributed; plus or minus 1=skewed and not normally distributed) to determine whether the data approximated a normal distribution (SAS Institute Inc. 1990). The kurtosis value of scenic beauty ratings, -0.5, suggested that rating distributions were normally distributed.

Nevertheless, we converted ratings to rankings for ease of interpretation and to minimize our assumptions. Ranked non-baseline image SBEz values served as the basis for conducting an analysis of variance by judge and treatment, calculation of averages, and F- and t-tests of significance at the 0.05 probability level. For brevity, quadrant differences are not included in this report. Statistical software used SAS's General Linear Model procedure (SAS Institute Inc. 1990). Examination of significant differences used the Bonferroni approach to ensure an experimentwise error rate by using t-tests at the 0.05/10=0.005 level (SAS Institute Inc.1990), and the Duncan multiple range (DMR) test (Cochran and Cox 1957) at the P=0.05 level.

RESULTS AND DISCUSSION

A visual inspection of color images suggested that the amount of sky in the scene and overt evidence of disturbance were negatively associated—and retained vegetation positively associated—with scenic beauty estimates. (A subset of the images with related SBE values is included in the appendix.) Scores ranged from –277.5 to +194.8, with a mean of –48.8 and a median of –58.8. Subsequent analyses used ranked scores ranging from 1 to 179, with a mean of 90.

Though there was considerable variation among the images, the analysis of variance revealed that significant (P(F)<0.05) variation in scenic beauty was due to differences among treatments (table 2). Each group of judges considered untreated stands highest and clearcut stands lowest in scenic beauty; other treatments were intermediate between these extremes (fig. 1). Overall average ranking for the informed forestry, senior forestry, and nonforestry classes who rated the images for scenic beauty were 92.0, 93.3, and 84.7, respectively. However, differences were not statistically significant by judge group (P(F=0.94, df = 2, 166)>0.39), or the treatment by judge group interaction (P(F=0.56, df=20, 146)>0.93).

Table 2—Analysis of variance of fall scenic beauty rating within a year following treatment, Ouachita-Ozark National Forests

Source	Degrees of freedom	Mean square variance	F value	P (larger F)
Treatment	10	25,008.51	17.45	< 0.001
Judge group	2	1,275.55	0.89	0.413
Judge by treatment	20	805.44	0.56	0.933
Residual	146	1,432.77		

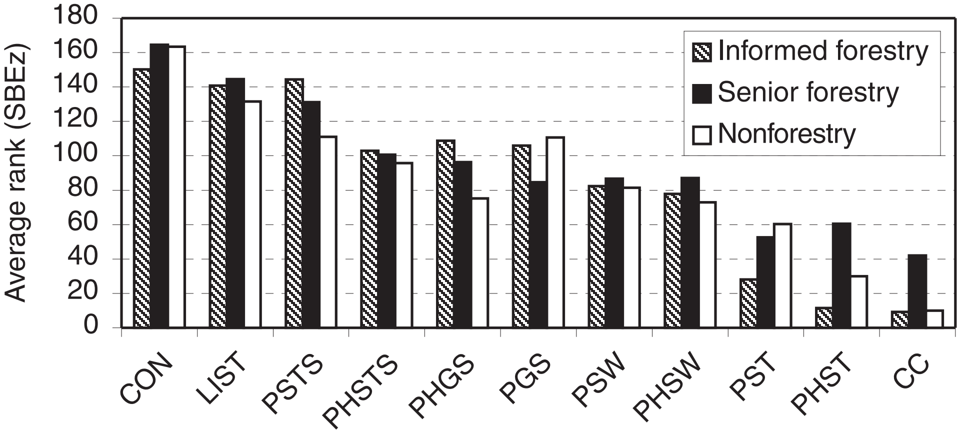

Figure 1—Average rank (standardized scenic beauty values) by judge group and year-earlier treatment for October 1993 shortleaf-hardwood near-stand views, Ouachita-Ozark National Forests. Treatments are listed in declining scenic beauty estimate rank order; CON = untreated stands; LIST = low-impact single-tree selection; PSTS = pine single-tree selection; PHSTS = pine-hardwood single-tree selection; PHGS = pine-hardwood group selection; PGS = pine group selection; PSW = pine shelterwood; PHSW = pine-hardwood shelterwood; PST = pine seed tree; PHST = pine-hardwood seed tree; CC = clearcut.

Treatments were the chief source of variation, with the pooled variance yielding P(F=18.4, df=10, 168)<0.001. Averages showed that untreated stands were rated and ranked highest, clearcut stands lowest, and less extensively cut stands intermediate between these two extremes (table 3). Treatments, grouped into similar impact categories and ordered by ascending negative impact, were: (1) low-impact and pine single-tree selection, (2) pine-hardwood single-tree selection, pine and pine-hardwood group selection, and pine and pine-hardwood shelterwood, (3) pine and pine-hardwood seed tree, and (4) clearcut harvest. Plotting the average ranking by the approximate square feet of basal area removed illustrates an inverse association (fig. 2).

Multiple comparison tests using the Bonferroni approach yielded the most conservative differences, as experiment-wise error rate for t-tests was set to P=0.005. With these test statistics, scenic beauty rankings of (1) untreated stands were indistinguishable from low-impact and pine single-tree selection; (2) group selection and shelterwood treatments were indistinguishable from each other; (3) pine and pine-hardwood treatments for a given cutting intensity were not significant; (4) seed tree cuts ranked lower than pine and low-impact single-tree selection and pine group selection; pine-hardwood seed tree cuts ranked still lower than pine group selection and pine-hardwood single-tree selection; and (5) pine and pine-hardwood seed tree cuts were indistinguishable from clearcuts. With Duncan's multiple range test, at P=0.05, smaller differences among closely ranked treatments were significant, but overall results remained the same.

CONCLUSIONS

Study results showed that timber cutting in the previous year negatively affected the scenic beauty of near-stand views in autumn. Differences among judge groups used for this study were not significant. We concluded that the use of different judge groups (from informed and uninformed forestry classes and a nonforestry class) had no significant influence on our findings. Our results corroborate Benson and Ullrich's (1981) suggestion that judge group differences (college students, public school teachers, and Forest Service researchers) had little effect on ratings of an array of treatments in Montana and Wyoming.

Scenic beauty of autumn, near-stand views were significantly lower and in inverse proportion to the amount of wood recently removed. Harvest intensity (approximated by square feet of basal area removed), was inversely associated with scenic beauty. Pine vs. pine-hardwood retention had no significant effect on scenic beauty for intensively cut stands. Overt signs of cutting dominated many of the scenes depicted

Table 3—Average scenic beauty estimate and average rank by year-earlier treatment, shortleaf-hardwood near-stand views, Ouachita-Ozark National Forests, October 1993

Treatment number and code	Rated images	Average SBEz	Average rank	DMR[b]	T-test results, P(|t|<0.005)[a] Treatment number 1	2	3	4	5	6	7	8	9	10	11
	no.														
1. CON	14	91.1	159.9	A	+	|	|	*	*	*	*	*	*	*	*
2. LIST	13	44.4	139.1	AB	|	+	|	|	|	|	|	*	*	*	*
3. PSTS	13	20.1	126.5	BC	|	|	+	|	|	|	|	|	*	*	*
4. PHSTS	26	- 34.0	99.4	CD	*	|	|	+	|	|	|	|	|	*	*
5. PHGS	18	- 43.1	92.2	D	*	|	|	|	+	|	|	|	|	|	*
6. PGS	13	- 29.6	102.1	CD	*	|	|	|	|	+	|	|	|	*	*
7. PSW	15	- 61.1	83.3	D	*	|	|	|	|	|	+	|	|	|	*
8. PHSW	28	- 65.7	79.9	D	*	*	|	|	|	|	|	+	|	|	*
9. PST	12	- 118.1	48.4	E	*	*	*	|	|	|	|	|	+	|	|
10. PHST	12	- 133.9	42.2	EF	*	*	*	*	*	|	|	|	|	+	|
11. CC	15	- 202.0	18.3	F	*	*	*	*	*	*	*	*	|	|	+
Overall	179	- 48.8	90.0												

SBEz = Scenic beauty estimate; CON = untreated stands; LIST = low-impact single-tree selection; PSTS = pine single-tree selection; PHSTS = pine-hardwood single-tree selection; PHGS = pine-hardwood group selection; PGS = pine group selection; PSW = pine shelterwood; PHSW = pine-hardwood shelterwood; PST = pine seed tree; PHST = pine-hardwood seed tree; CC = clearcut.

[a] T-tests (Bonferroni approach): compared with treatment (+) averages, other averages were significantly (*) different at alpha = 0.005, (|) otherwise.

[b] Duncan's Multiple Range test: averages with the same letter are not significantly different, (P < 0.05).

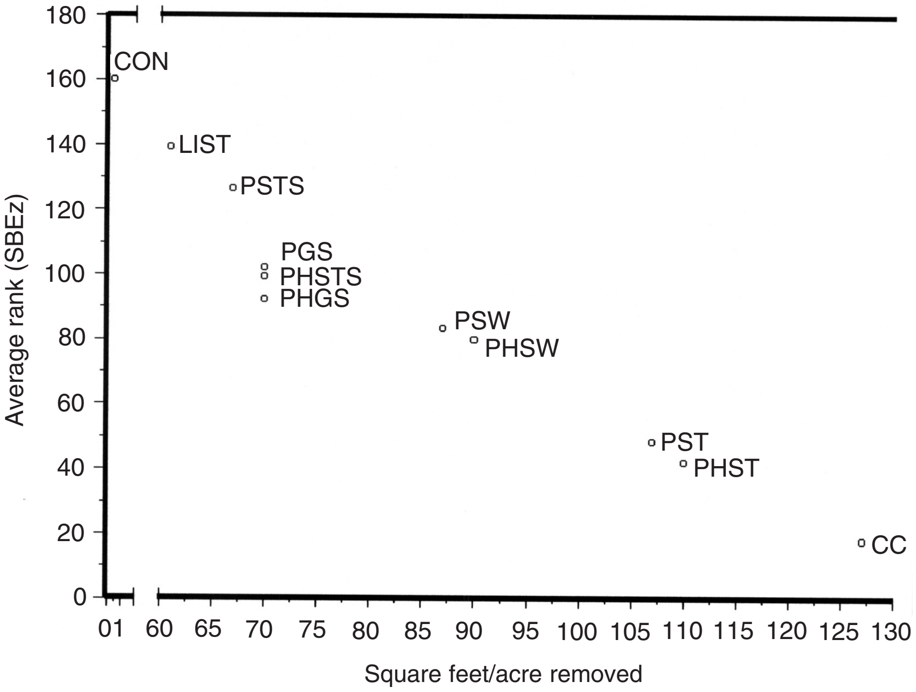

Figure 2—Average scenic beauty rank by approximate square feet of basal area removed in year-earlier treatments for October 1993 shortleaf-hardwood near-stand views, Ouachita-Ozark National Forests; CON = untreated stands; LIST = low-impact single-tree selection; PSTS = pine single-tree selection; PHSTS = pine-hardwood single-tree selection; PHGS = pine-hardwood group selection; PGS = pine group selection; PSW = pine shelterwood; PHSW = pine-hardwood shelterwood; PST = pine seed tree; PHST = pine-hardwood seed tree; CC = clearcut.

in near-stand images of treated stands and may have overwhelmed the finer-scaled differences in pine vs. pine-hardwood retention, and within-stand grouped vs. single-tree cutting arrangements. We further concluded that cutting alternatives fell into four groups according to impact. These were, in order of increasing impact on scenic beauty: (1) low-impact and pine single-tree selection, (2) pine-hardwood single-tree selection, group selection, and shelterwood, (3) seed tree, and (4) clearcut treatments.

Whether the relationships among treatments suggested by our results remain the same for other stand types, within-stand views, or different seasons is uncertain. With foreground views of Montana Douglas fir stands, studies by Benson and Ullrich (1981) yielded scenic beauty ratings for shelterwood significantly greater than clearcut stands. Further refinement could include forecasting the decline in near-stand scenic beauty of shortleaf-hardwood stands as a function of the actual amount of wood removed, rather than the approximate values provided in this report. With the existing images, future image analysis might reveal color (evergreen pine vs. hardwood) differences between pine and pine-hardwood-single-tree retention, and discrimination of image attributes among the four groups of treatment impacts noted above.

Forest managers concerned with public perception of harvests need to consider near-stand scenic beauty impacts when choosing harvest regimes, particularly along roadside views and in public forests. Our study suggests that—within a year of treatment—single-tree selective cutting has the least impact, followed by shelterwood, seed tree, and clearcut treatments. Pine single-tree selective cutting has less of an impact than pine-hardwood group selection and shelterwood cutting. Shelterwood stands have greater scenic beauty than seed tree and clearcut stands; pine seed tree stands have greater scenic beauty than clearcut stands. Our study suggests that the spatial arrangement or proportion of hardwood retained may not influence scenic beauty of near-stand views as much as the basal area of trees removed.

Nevertheless, vegetation structure recovers from harvests, trees regenerate, and scenic beauty improves in the years following treatment, as shown for within-stand views of uneven-aged (Gritter 1997, Rudis and others 1999) and even-aged treatments (Rudis and others, in press) and foreground views of a variety of treatments (Benson and Ullrich 1981). For near-stand views, we suspect that harvest disturbance impacts become less obvious and scenic beauty improves with the passage of time.

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APPENDIX

The following 12 photographs (six untreated and six clearcut) is a subset of the study's digitally archived images with identifying codes by treatment and stand. Examples are ordered by type of treatment and scenic beauty scores within each treatment. References under each image are the 6-digit stand and compartment code, the 7-digit original compact disk and image code, and the standardized scenic beauty estimate (SBEz).

Untreated

Stand 0284-11 CD 0677-066
SBEz 194.80

Stand 0023-10 CD 0329-017
SBEz 155.45

Stand 0023-10 CD 0329-015
SBEz 134.01

Stand 0284-11 CD 0677-060
SBEz 99.35

Stand 0605-05 CD 0677-018
SBEz 62.40

Stand 0605-05 CD 0677-020
SBEz 19.91

Clearcut


Stand 1067-15 CD 0677-084
SBEz -92.56


Stand 1685-05 CD 0329-061
SBEz -159.75


Stand 1658-05 CD 0329-060
SBEz -212.12


Stand 1067-15 CD 0677-083
SBEz -237.76


Stand 0458-16 CD 0677-028
SBEz -260.24


Stand 0458-16 CD 0677-029
SBEz -277.51

A PSYCHOLOGICAL MODEL OF SCENIC BEAUTY BY SILVICULTURAL TREATMENT TWO GROWING SEASONS AFTER HARVEST

Ying-Hung Li, Victor A. Rudis, and Theresa A. Herrick[1]

Abstract—This study estimated summer scenic beauty and associated psychological attributes of scenes depicting uncut and several cutting regimes within shortleaf pine-hardwood forests on national forests. Images were captured in the summer of 1994 in nine treated and three comparable untreated stands in the Ouachita Mountains of Arkansas. Treatments imposed in the winter of 1992-93 included group selection, pine-hardwood shelterwood, and clearcut in north, east, and south quadrants of the region. Landscape Architecture professionals, students with professional training, and other students with no training, rated scenic beauty preferences and associated psychological attributes. Analysis of rankings showed significant differences (P(F)<0.05) in psychological attributes by treatment type and the background of judges. For all judges, more intensive cutting yielded significantly less scenic beauty, mystery, coherence, and complexity, and greater visual penetration. Legibility, a term used to describe finding one's way, was not significantly associated with cutting treatment. Scenic beauty preferences were indistinguishable among intermediate (shelterwood and group selection) treatments, although group selection was likely the least offensive because it provided mystery, complexity, and visual penetration comparable to untreated areas. There were significant quadrant-by-treatment interactions, suggesting that local conditions also affect the impact of treatments on scenic beauty. Our results lend quantitative credence to the qualitative notion that adapting cutting practices to limit visual penetration and increase coherence, complexity, mystery, and scenic beauty can yield measurable aesthetic benefits.

INTRODUCTION

The National Environmental Policy Act of 1969 requires environmental impact analyses for major projects on Federal land, including assessments of aesthetics. In general, older timber stands and those with open, park-like settings and limited understory vegetation are preferred; young stands and the presence of abundant logging slash from recent cutting are not (Ribe 1989). While the negative impact of timber harvests on scenic beauty of forested land is well known (Benson and Ullrich 1981, Jones 1993, Ribe 1989, Vodak and others 1985), it is poorly understood.

Any scientific assessment of aesthetics uses approaches rooted in psychology or psychophysics (Daniel and Vining 1983). Psychophysics studies the relationship between physical objects and the aesthetic preference of respondents. Through measurement of physical objects, such as the number of tree stems, and a series of preference ratings, the scientist can develop models that relate objects to preferences. Models developed with physical objects tied to psychological theory, such as visual penetration, constitute psychophysical approaches (e.g., Ruddell and others 1989, Rudis and others 1988). Psychological studies emphasize theoretical constructs for the aesthetic response, which describes both how people perceive (make sense of) and organize visual information and how previous experience influences their aesthetic response.

On public land, visual landscape management by U.S. agencies employs the expertise of landscape designers, as well as empirical public preference research (Anderson 1995). For managed forests, empirical aesthetics research focuses on public judgments, largely based on scenic beauty preferences. Yet landscape designers make use of a wider array of psychological attributes than scenic beauty, such as mystery and coherence. Most attributes have been empirically studied only within urban environments (Kaplan and Kaplan 1989). Examination of psychological attributes in managed forested environments should help us understand the causes behind public preference for particular scenes and forest management regimes.

Though landscape designers might not always use the same terminology, most agree that how one perceives a scene involves an array of organizing principles (Motloch 1991). In psychological terms, Kaplan and Kaplan (1989) theorized that one's preference for particular natural scenes was evoked by information processing components, namely coherence, complexity, mystery, and legibility (finding one's way in a scene). Based on Ruddell and others' (1989) psychophysical modeling, Li and Hammitt (1999) added visual penetration as another information-processing component. A theoretical causal model for scenic beauty preferences includes these five components as principal causal factors affecting scenic beauty preferences. If these factors control perception, then comparison of causal attributes by cutting practice should reveal the psychological factors that determine perceived changes in scenic beauty.

OBJECTIVES AND STUDY AREA

Our overall objective was to better understand the psychological response to timber cutting practices on national forests of the Ouachita Mountains. Specific objectives were threefold:

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1. To test the effect of landscape architecture knowledge differences among groups of judges on scenic beauty ratings
2. To test operational-scale cutting practices on scenic beauty of shortleaf pine (*Pinus echinata* Mill.)-hardwood stands, and as a basis for understanding scenic beauty ratings
3. To test operational-scale cutting practices on ratings of causal psychological attributes (coherence, complexity, legibility, mystery, and visual penetration).

Study locations were part of a 52-stand study region on national forest land in the Ouachita Mountains, an area 120 miles east-to-west, 80 miles north-to-south, and located in northwest Arkansas and eastern Oklahoma (Baker 1994, Guldin and others 1994). For operational purposes, the 9,600 square mile study area was divided into quadrants. Quadrants and nearby cities were: north (Danville, AR), east (Cedar Creek, AR) south (Mount Ida, AR), and west (Black Fork, AR). Trees were cut largely to promote pine reproduction. Treatments were harvests in the winter of 1992-93 and site preparation for regeneration early in 1994. The study was designed as a randomized complete block, with silvicultural treatments randomly assigned to 1 of 13 stands in each of the quadrants (Baker 1994). Because of funding limitations, we had limited personnel. In taking photographs, we also sought to minimize variability due to within-season weather conditions. For these reasons, we restricted our examination to the north, east, and south quadrants of the study region.

The north quadrant is in the Arkansas Valley ecoregion, with the other quadrants in the midst of the Ouachita Mountain ecoregion, where streams tend to be ephemeral, and in the south quadrant where they tend to be permanent (Baker 1994, Guldin and others 1994). Stands in the north, east, and south quadrant have average slopes of 15.1, 12.3, and 9.7 percent; 50-year shortleaf site index values of 61.5, 62.2, and 65.8 feet; average micro relief severity ratings (1 mild, 2 moderate, 3 severe) of 1.6, 1.5, and 1.4; and tree ages of 65.4, 65.3, and 62.7 years, respectively (Guldin and others 1994).

Cost and time constraints limited this study to four treatment conditions (National Forest stand and compartment number):

a. Untreated: untreated stands retained in their natural state, referenced commonly in the design of experiments as a control (CON), averaging 130 cu ft/ac. (The north, east, and south stand compartments were 0284-11, 0605-05, and 0023-10, respectively.)

b. Group selection: pine-hardwood group selection (PHGS): clearcut 0.1 to 2.0 ac. Retention of 20-to-45 trees/ac shortleaf pine and 10-to-20 trees/ac hardwoods. (The north, east, and south stand compartments were 0046-18, 1124-11, and 0035-42, respectively.)

c. Shelterwood: pine-hardwood shelterwood (PHSW): Retention of 10-to-30 trees/ac shortleaf pines and 10-to-30 trees/ac hardwoods. (The north, east, and south stand-compartments were 0457-12, 1119-21, 0027-01, respectively.)

d. Clearcut: all merchantable volume removed (CC). (The north, east, and south stand compartments were 0458-16, 1067-15, and 1658-05, respectively.)

METHODS

Approximately 15 photographic images per stand were acquired in July 1994 using 35-mm ASA 400 transparency film, push-processed to ASA 800, and an f2.8 lens (Olympus XA) from 5 or 6 point locations stratified within stands that were designated for periodic bird censuses. Images were acquired two growing seasons after harvest disturbance, generally on sunny-to-partly-cloudy days not before 7:30 AM and not after 6:00 PM. Duplicate, over- or underexposed images, and those with human subjects, were discarded. What remained were images from which random samples were selected by treatment. Random sample selection yielded one to four images representing a single stand, with all but two stands represented by three images.

Judges first viewed all 36 images for 2 seconds each to give them a feel for the range of variation. Then, after being shown each scene for 8 seconds, they rated each image on a scale from 1 (lowest) to 10 (highest). This numerical indication of people's perception is used widely in scenic beauty estimation (Brown and Daniel 1990; Daniel and Vining 1983, Ribe 1989), and all follow procedures outlined by Daniel and Boster (1976).

There were 88 judges—all from Clemson University. Of these, 70 were nonprofessionals, i.e., students from two undergraduate introductory courses (one in psychology, the other in horticulture), 14 were professionals, i.e., students from two upper-level landscape design studio classes, and 4 were experts, i.e., professional educators or researchers familiar with landscape preference research. The judges were 56 percent female and 44 percent male. Rating sessions were held in the spring of 1996 during class periods.

To avoid participant fatigue in rating the images, we used two rating sessions. In one session, judges coded scenic beauty, coherence, and legibility. In the second session a week later, judges coded complexity, mystery, and visual penetration. To avoid conditioning of responses to identical stimuli, participants rated images one attribute at a time. Image order was randomly assigned for each psychological attribute. That is, the sequence of images for scenic beauty differed from the sequence for coherence and for legibility.

We asked participants to rate images but gave them no instruction about what constituted scenic beauty. We then instructed them in the use of causal psychological attributes and showed them images suggesting high, medium, and low values for each psychological attribute. Definitions presented were:

> Coherence—how easy it is to visually organize the trees and surrounding vegetation into a well-ordered pattern, and how well the patterns "hang together"
>
> Complexity—how much there is to look at in this forest scene in terms of the number of different kinds of trees, clusters of trees, and other vegetation compositions or natural elements

Legibility—how easy it would be for them to find their way around in the forest area using noticeable landmarks such as big trees, clusters of trees, unusually shaped trees or bushes, clearings, pathways, or any other memorable features

Mystery—how much this forest scene attracts and encourages them to go beyond their standing-viewing point, enter deeper into the forest, and see things that are only hinted at from their current position.

Visual Penetration—the ability to see through a forest scene without interruption by vegetation screening, tree trunks, or other noticeable objects.

To focus on stated objectives and simplify analysis, we averaged ratings for each of the 36 images by judge group (expert, professional, nonprofessional), quadrant (north, east, south), and treatment (untreated, group selection, shelterwood, clearcut). In performing statistical tests, we weighted average values to reflect the number of judges represented.

For statistical analytical purposes, we assumed the resulting ordinal scale preference ratings were interval data, with an implied uniform distance between two adjacent ratings. This is a widely accepted assumption in social science preference studies and commonly is used in scenic beauty preference research (Daniel and Vining 1983). The reader is cautioned, however, that in fact no true interval exists. That is to say, a difference of 1.0 between 5.0 and 6.0, does not necessarily represent the same difference between 9.0 and 10.0.

Traditional parametric tests assume an underlying normal distribution, but ratings may not follow such a distribution. We tested the distribution of average scenic beauty ratings for kurtosis (0=not skewed and normally distributed; plus or minus 1=skewed and not normally distributed) to determine whether the data approximated a normal distribution (SAS 1990). The kurtosis value for scenic beauty, -0.9, suggested rejection of the null hypothesis that the distribution was normally distributed.

Ratings for all psychological attributes were converted to rankings to conduct an analysis of variance (ANOVA) and calculation of F (equivalent to nonparametric) tests of significance at the 0.05 probability level. Analyses were conducted with average ranked values, but averages are reported on a 10-point rating scale for ease of interpretation. Statistical software employed SAS's General Linear Model (GLM) procedures (SAS 1990). Multiple comparison tests used the Bonferroni approach to ensure an experimentwise error rate of 0.05 by using t-tests each at the 0.05/3=0.017 level with SAS's least significant means (LSMEANS), GLM, and mixed ANOVA (MIXED) procedures (SAS 1990, 1996).

RESULTS

A visual inspection of color images suggested that the amount of sky in a scene and overt evidence of disturbance were negatively associated with scenic beauty estimates, and retained vegetation was positively associated. Each group of judges rated untreated stands highest in scenic beauty and clearcut stands lowest. Other treatments were intermediate between these extremes. Black-and-white versions of the images are included in the appendix, along with an average rating for each psychological attribute.

For all quadrants, untreated stands were rated highest and clearcut stands lowest. Most of the variation was among treatments, although the ANOVA revealed that perceived scenic beauty was significantly affected by the interaction between quadrant and treatment (table 1). Nevertheless, by quadrant, clearcut stands were rated lowest and untreated stands highest (fig. 1).

There was no significant interaction between judges and treatments (*P*(larger F)=0.259). F-tests revealed significant differences in perceived scenic beauty among judge groups (*P*(larger F)=0.002). So regardless of treatment, scenic beauty ratings from the expert, professional, and nonprofessional judges were 4.7, 5.8, and 5.5, respectively. Experts gave significantly different (*P*|t|<0.01) and consistently lower rankings, but differences between judges with professional and nonprofessional architecture backgrounds did not (*P*|t|=0.03) (fig. 2).

Other psychological attributes most closely aligned with scenic beauty were, in order of association: mystery, coherence, visual penetration, complexity, and legibility (table 2). Mystery and coherence were the two psychological attributes most closely associated with scenic beauty. Judges gave untreated stands the highest average value for scenic beauty, mystery, coherence, and complexity, and lowest average values for legibility and visual penetration. For brevity, differences in other psychological attributes by

Table 1—Analysis of variance of scenic beauty ranking two growing seasons after treatment, Ouachita Mountains

Source	Degrees of freedom	Mean square variance	F value	*P* (larger F)
Quadrant	2	1,373.01	0.16	0.858
Treatment	3	26,462.82	9.39	0.009
Quadrant by treatment	6	8,802.82	50.53	< 0.001
Judge	2	1,577.11	9.05	0.002
Judge by treatment	6	251.74	1.44	0.259
Residual	16	174.22		

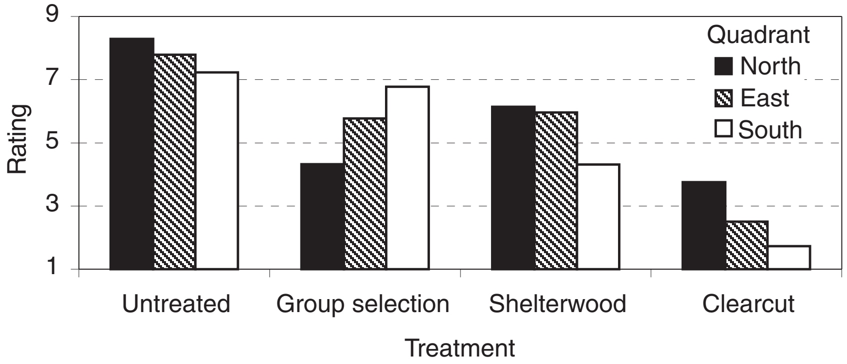

Figure 1—Average scenic beauty ratings two growing seasons after treatment, Ouachita Mountains, by treatment and quadrant of the study region.

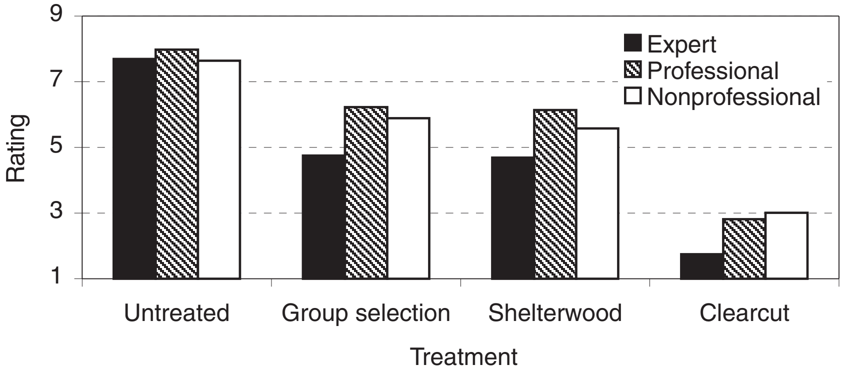

Figure 2—Average scenic beauty ratings two growing seasons after treatment, Ouachita Mountains, by judges with different landscape architecture backgrounds.

Table 2—Correlation between scenic beauty and other psychological attribute rankings (n=36) two growing seasons after treatment, Ouachita Mountains[a]

Code and attribute name	SBE	MY	CH	CM	LE
Visual penetration	-0.76	-0.93	-0.91	-0.93	0.86
Legibility	-0.52	-0.86	-0.71	-0.75	1.00
Complexity	0.59	0.83	0.58	1.00	—
Coherence	0.91	0.87	1.00	—	—
Mystery	0.93	1.00	—	—	—

SBE = Scenic beauty estimate; MY = mystery; CH = coherence; CM = complexity; LE = legibility.

[a] All have significant Pearson correlation coefficients (*P*Ilarger rI<0.001).

quadrant and by the landscape architecture background of judges are not included in this report.

Apart from scenic beauty, statistically significant (*P*(larger F)<0.05) differences by treatment and causal psychological attribute were for mystery, coherence, complexity, and visual penetration, but not legibility (fig. 3). By treatment, group selection was not significantly different from untreated stands for any of the causal attributes. Shelterwood treatments resembled untreated stands only in coherence and legibility. Clearcut treatments resembled untreated areas only in legibility.

DISCUSSION

Our study showed that of the three treatments, clearcut stands were rated lowest in scenic beauty. Alhough there was no significant difference in scenic beauty ratings of shelterwood and group selection treatments, examination of causal attributes revealed differences between these two in complexity and visual penetration.

Our study was designed to represent the Ouachita Mountains study region, and we had not anticipated a large and significant quadrant by treatment effect, but our results showed otherwise. Guldin and others (1994) documented significant differences in topography by quadrant, with gentler to steeper slopes and lesser to greater micro relief severity from south to east to north. Distant vistas afforded by shelterwood cutting could well have influenced scenic beauty ratings more in the north than in the south. The south quadrant was more mesic (i.e., higher site index) and younger, on average, than the north; so more abundant logging slash remained visible there than in the other quadrants. Our results yielded lower averages for clearcut stands, regardless of quadrant. This is consistent with Vodak and others

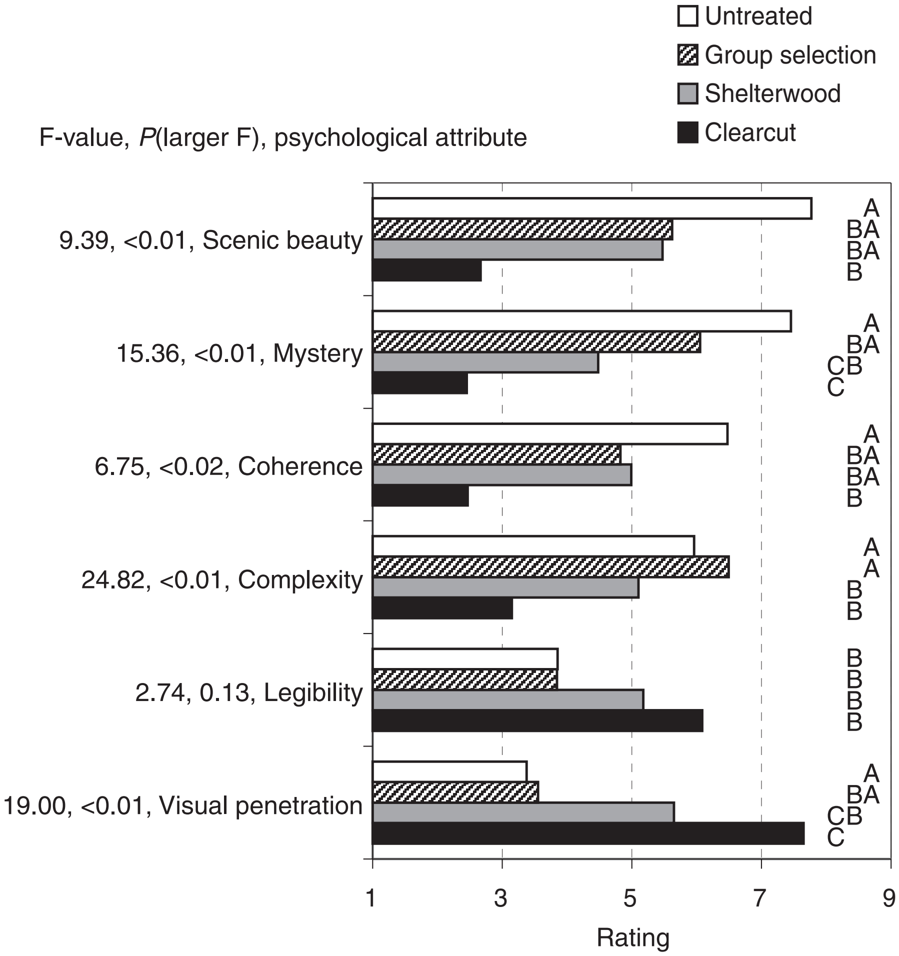

Figure 3—F-value, *P*(larger F) for the treatment effect by psychological attribute, and average rating for summer scenes two growing seasons after treatment, Ouachita Mountains. Averages by attribute with the same letter are not significantly different, using least squares means tests for multiple comparisons (*P*|t|<0.01).

(1985) study that found recent and abundant logging slash detracting from scenic beauty.

Obviously, the expert judges had more experience with landscape evaluation than professional and nonprofessional judges. We anticipated that experts would be more sensitive to differences in scenic beauty, and professionals more sensitive than nonprofessionals but less sensitive than experts. We found only that experts provided lower ratings but otherwise were no different from other groups in their comparative rating of scenic beauty by treatment. For the particular mix of individuals in our sample, none of the three groups of judges rated group selection and shelterwood treatments differently. Repeated experimentation is needed to verify such results, because our sample of both experts (n=4) and professionals (n=14) was limited.

The inclusion of various causal attributes provided insight into psychological reasons for scenic beauty preferences. Overall, group selection or shelterwood were favored over clearcut treatments. Our findings also corroborate an earlier plot-level study of within-stand scenes on north-facing slopes 2 years after treatment. On the Winona Ranger District, Gramann and Rudis (1994) showed that group selection was the favored treatment among other uneven-aged reproduction cutting regimes. They also showed views of north-facing, ridgetop views were more scenic, on average, than views on more mesic, gentler slopes.

Mystery and coherence were most closely correlated with scenic beauty. In a related study using structural equation modeling (SEM), Li and Hammitt (1999) tested a modified version of Kaplan and Kaplan's (1989) information processing components. SEM, widely used in the social sciences (Byrne 1994), is analogous to factor analysis, but with selection of abstract factors constrained by a theoretical, causal construct. Factor analysis uses an unconstrained selection of factors based on an analysis of covariance, i.e., the relationship of measured attributes to each other and upon the dependent attribute of interest. SEM, in brief, incorporates theoretical attributes that may not be directly measured. Further details are provided elsewhere (Byrne 1994, Li and Hammitt 1999).

Li and Hammitt (1999) found that most (81 percent) of the variance in perceived scenic beauty among expert judges varied with two theoretical factors: one factor directly associated with coherence and mystery; and a second factor directly associated with visual penetration and legibility and inversely associated with complexity. Standardized coefficients were 0.91 for the first factor and –0.08 for the second factor, which suggested the first factor was the chief theoretical cause for scenic beauty preferences.

Our direct study of individual causal attributes shows group selection having the most mystery and complexity, and the least amount of visual penetration of all treatments, and not substantially different from untreated areas. Legibility was not a useful metric to distinguish among treatments. We suggest that digital methods to analyze these scenes (Kalidindi and others 1996, Rudis and others 1999) should be helpful in assessing image metrics or features that reflect these and other theoretically relevant psychological attributes.

Few physical attributes have been directly associated with psychological measures, other than scenic beauty and visual penetration. Our findings differ from Rudis and others (1988) and Ruddell and others (1989) regarding visual penetration. Their studies involved an examination of 99 east Texas loblolly pine (*Pinus taeda* L.)-shortleaf pine (*P.echinata* Mill.) and oak (*Quercus* spp.) sample locations across five counties, largely on private land. Our study involved only 12 stands located in shortleaf pine-hardwood stands located in National Forests of the Ouachita Mountains of Arkansas, and by design, three-fourths of the scenes were of 2-year-earlier cutting activities. Our study had many more scenes of more recently disturbed vegetation than the east Texas scenes. Their "visual penetration" also referenced an ocular, scaled physical measurement of visual penetration, rather than the visual penetration perceived by judges.

MANAGEMENT IMPLICATIONS

Aside from approaches to minimize the aesthetic impacts of timber harvesting (Jones 1993), the goals of traditional cutting and management are to maximize timber yield while minimizing damage to the residual stand. The primary goal focuses on the selection of merchantable trees and the status of individual remnant trees, rather than the spatial arrangement of gaps created in the forest or their association with the surrounding ecoregion. Our study lends quantitative credence to the notion that there is a measurable benefit to adapting cutting and management regimes that foster scenic beauty.

Landscape architects commonly design landscapes by organizing elements to promote mystery and coherence, e.g., by making winding paths, small openings, and focal points and creating smooth transitions among objects with similar form (e.g., Motloch 1991). Our study demonstrates that standard, operational-scale cutting regimes do affect mystery and coherence within the scenes changed by traditional logging operations. To retain scenic beauty while harvesting timber, intermediate cutting practices (group selection and shelterwood) are least offensive. The choice between group selection and shelterwood will depend on characteristics within the ecoregion.

Although we can only speculate on the utility of our findings for other types of logging operations and other ecoregions, we offer quantitative credence to the psychological impact of timber harvesting. Adapting cutting practices to novel cutting regimes that increase mystery, coherence, complexity, and scenic beauty, while reducing visual penetration may even foster public acceptance of harvest operations.

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APPENDIX

Within each condition, image order is from highest to lowest scenic beauty value.
The legend for each image is:

Treatment condition
Stand: National Forests in Arkansas, stand and compartment code;
Pt: Point number;
Azimuth: compass direction of view when standing at the point:
CD: compact disk and image number.
Psychological attributes, each scaled 1 (lowest) to 10 (highest):

Code	Meaning	Code	Meaning
SBE	Scenic beauty estimate	CH	Coherence
LE	Legibility	CM	Complexity
MY	Mystery	PV	Visual Penetration

Untreated

Stand 0284-11 Pt 2 Azimuth 220
CD 0015-024

SBE	8.4	CH	9.0
LE	4.9	CM	3.7
MY	6.6	PV	5.2

Stand 0284-11 Pt 6 Azimuth 150
CD 0015-030

SBE	8.1	CH	5.8
LE	2.4	CM	6.9
MY	8.0	PV	3.0

Stand 0284-11 Pt 3 Azimuth 360
CD 0015-022

SBE	8.1	CH	6.9
LE	2.9	CM	6.6
MY	8.4	PV	2.6

Stand 0605-05 Pt 5 Azimuth 270
CD 2607-068

SBE	7.7	CH	6.5
LE	6.1	CM	5.3
MY	6.5	PV	4.1

Stand 0605-05 Pt 2 Azimuth 120
CD 2607-075

SBE	7.7	CH	6.5
LE	3.2	CM	6.7
MY	7.5	PV	3.0

Stand 0605-05 Pt 4 Azimuth 120
CD 2607-064

SBE	7.7	CH	7.2
LE	4.4	CM	5.6
MY	7.2	PV	3.8

Stand 0023-10 Pt 3 Azimuth 180
CD 2607-007

SBE	7.3	CH	5.2
LE	3.7	CM	6.5
MY	7.2	PV	3.1

Stand 0023-10 Pt 6 Azimuth 150
CD 2607-013

SBE	7.2	CH	3.6
LE	2.9	CM	7.6
MY	6.3	PV	2.1

Stand 0023-10 Pt 6 Azimuth 360
CD 2607-015

SBE	6.9	CH	5.2
LE	3.8	CM	5.8
MY	6.3	PV	2.9

Group Selection

Stand 0035-42 Pt 4 Azimuth 300
CD 2607-084

SBE	7.1	CH	5.7
LE	2.0	CM	7.5
MY	7.6	PV	2.6

Stand 0035-42 Pt 3 Azimuth 180
CD 2607-082

SBE	7.0	CH	4.9
LE	2.1	CM	7.6
MY	7.7	PV	1.9

Stand 0035-42 Pt 2 Azimuth 120
CD 2607-081

SBE	6.9	CH	5.2
LE	6.0	CM	7.8
MY	6.9	PV	3.5

Stand 1124-11 Pt 1 Azimuth 240
CD 2607-025

SBE	6.6	CH	6.4
LE	4.4	CM	5.7
MY	7.1	PV	3.5

Stand 1124-11 Pt 4 Azimuth 120
CD 2607-028

SBE	6.5	CH	7.9
LE	5.3	CM	4.2
MY	5.8	PV	4.9

Stand 0046-18 Pt 1 Azimuth 240
CD 0012-050

SBE	5.8	CH	2.8
LE	7.0	CM	7.5
MY	5.8	PV	2.7

Stand 0046-18 Pt 5 Azimuth 270
CD 0012-57

SBE	5.5	CH	4.2
LE	5.1	CM	7.2
MY	5.0	PV	4.1

Stand 1124-11 Pt 3 Azimuth 360
CD 2607-030

SBE	4.8	CH	2.6
LE	3.2	CM	7.6
MY	5.9	PV	2.3

Stand 0046-18 Pt 2 Azimuth 210
CD 0012-052

SBE	2.3	CH	3.9
LE	5.6	CM	4.6
MY	4.0	PV	5.3

Shelterwood

Stand 0457-12 Pt 2 Azimuth 030
CD 2607-048

SBE	8.4	CH	6.4
LE	6.9	CM	4.3
MY	4.2	PV	6.7

Stand 1119-21 Pt 3 Azimuth 180
CD 0012-095

SBE	7.0	CH	5.5
LE	5.3	CM	5.4
MY	4.7	PV	5.0

Stand 1119-21 Pt 2 Azimuth 120
CD 0012-090

SBE	7.0	CH	5.0
LE	3.2	CM	5.1
MY	5.0	PV	4.8

Stand 0027-01 Pt 5 Azimuth 090
CD 2607-058

SBE	5.2	CH	4.9
LE	2.1	CM	5.6
MY	3.1	PV	5.9

Stand 0457-12 Pt 4 Azimuth 300
CD 2607-040

SBE	5.2	CH	5.1
LE	5.6	CM	5.3
MY	5.3	PV	4.5

Stand 0457-12 Pt 4 Azimuth 120
CD 2607-039

SBE	5.2	CH	4.1
LE	5.1	CM	5.9
MY	4.9	PV	5.2

Stand 0027-01 Pt 2 Azimuth 030
CD 2607-050

SBE	4.2	CH	3.5
LE	3.6	CM	5.7
MY	4.3	PV	5.4

Stand 1119-21 Pt 1 Azimuth 240
CD 0012-094

SBE	4.2	CH	5.2
LE	4.4	CM	5.1
MY	4.6	PV	5.5

Stand 0027-01 Pt 6 Azimuth 330
CD 2607-060

SBE	3.9	CH	3.3
LE	2.0	CM	5.2
MY	3.4	PV	6.3

Clearcut

Stand 0458-16 Pt 6 Azimuth 330
CD 0015-009

SBE	4.2	CH	3.0
LE	5.4	CM	4.2
MY	2.6	PV	6.5

Stand 0458-16 Pt 2 Azimuth 210
CD 0015-002

SBE	4.0	CH	2.6
LE	6.9	CM	4.4
MY	3.5	PV	6.3

Stand 1067-15 Pt 3 Azimuth 360
CD 0012-078

SBE	3.5	CH	3.4
LE	6.5	CM	4.2
MY	3.8	PV	6.8

Stand 1067-15 Pt 5 Azimuth 090
CD 0012-066

SBE	3.2	CH	2.4
LE	5.9	CM	3.2
MY	2.5	PV	7.3

Stand 1067-15 Pt 5 Azimuth 270
CD 0012-067

SBE	2.9	CH	2.8
LE	6.2	CM	4.0
MY	2.6	PV	7.0

Stand 1658-05 Pt 3 Azimuth 180
CD 0012-034

SBE	2.7	CH	2.2
LE	6.9	CM	3.3
MY	2.5	PV	7.4

Stand 1658-05 Pt 5 Azimuth 270
CD 0012-043

SBE	1.9	CH	2.3
LE	8.0	CM	2.3
MY	1.6	PV	8.8

Stand 1067-15 Pt 5 Azimuth 030
CD 0012-068

SBE	1.8	CH	1.7
LE	7.0	CM	2.8
MY	2.1	PV	8.6

Stand 1658-05 Pt 2 Azimuth 210
CD 0012-037

SBE	1.6	CH	1.6
LE	6.4	CM	2.7
MY	1.9	PV	8.5

Session 6
Phase III Overview and Wildlife Research

T. Bently Wigley, Moderator

LANDSCAPE-SCALE RESEARCH IN THE OUACHITA MOUNTAINS OF WEST-CENTRAL ARKANSAS: GENERAL STUDY DESIGN

James M. Guldin[1]

Abstract—A landscape-scale study on forest ecology and management began in 1995 in the eastern Ouachita Mountains. Of four large watersheds, three were within the Winona Ranger District of the Ouachita National Forest, and a major forest industry landowner largely owned and managed the fourth. These watersheds vary from 3,700 to 9,800 acres. At this scale, replicating treatments for statistical uniformity is impossible; this situation requires other approaches to statistical design. One approach is temporal replication, which uses multiple years of baseline measurement both prior to and after treatment as a source of experimental error and hypothesis testing. The second develops models that can be tested using data subsets, or the development of additional reference watersheds for testing models.

INTRODUCTION

The phase III landscape study in the Ouachita Mountains Ecosystem Management Research Project was conceived as a way to deal with questions about forest management that cannot be answered at a stand-level scale, but that can be answered in the context of a watershed that contains many stands (Hornbeck and Swank 1992, Kessler and others 1992, Lubchenko and others 1991, National Research Council 1990, O'Hara and others 1994, Swanson and Franklin 1992). These questions include (1) cumulative management effects on perennial streams, including both hydrology and aquatic ecology; (2) the behavior of organisms whose home range encompasses multiple stands or watersheds; and (3) the degree of change in vegetation in a watershed as a result of a combination of management activities in a subset of the stands in that watershed.

The objectives of research in the phase III study are to

(1) quantify core watershed hydrology through modeling hydrological factors and cumulative hydrological effects, using a series of flumes and uncontrolled cross-section gauging stations

(2) characterize and quantify sensitive and critical elements of aquatic and riparian ecology

(3) quantify terrestrial ecological relationships of vegetation pattern, ecological classification, wildlife, and biodiversity at the landscape scale

(4) characterize the social dimensions of the landscape, including the prehistoric, historic, and current relationships of people with the land.

To achieve these objectives, four watersheds were selected to represent a sequence of initial forest management conditions, from relatively unmanaged to intensively managed.

METHODS

Study Area

Three of the four watersheds are east of State Highway 7 on the Winona Ranger District in the combined Jessieville/Winona Ranger Districts of the Ouachita National Forest in Saline County, AR. These three contiguous watersheds are part of the Upper Lake Winona drainage basin, from which water flows eastward via Alum Creek and its two tributaries, the North Alum Creek and Bread Creek, into Lake Winona. Ultimately, those waters flow southerly to the Saline River of central Arkansas.

The fourth watershed, roughly 10 miles southwest of the others, is west of Highway 7 on land owned and managed by Weyerhaeuser Company. Water flows via the Little Glazypeau Creek into the Ouachita River, below Lake Ouachita but above Lake Hamilton west of Hot Springs.

The Ouachita and Saline Rivers meet in south Arkansas in Felsenthal National Wildlife Refuge. Thus, although the four watersheds are in close proximity and share a similar climatic regime, there are ecological differences between the Ouachita and Saline river systems that might affect comparisons among the watersheds.

Alum Creek Watershed

The Alum Creek watershed (3,700 acres), the unmanaged control in the study, was imposed largely on the Alum Creek Experimental Forest in compartments 1457 and 1460 of the Winona Ranger District. The Government owns virtually all of this watershed. The Alum Creek Experimental Forest has been largely used for upland small-catchment hydrology research. As a result, most of the experimental forest has had no management for two decades. The boundary of this watershed was established using a digital elevation model, which drew the hydrological unit of interest slightly outside the current boundary of the Alum Creek Experimental Forest.

Bread Creek Watershed

The Bread Creek watershed (3,800 acres), east of the Alum Creek watershed, includes compartment 1462 and the south half of compartment 1456 of the Winona Ranger District. The Government owns 95 percent of this watershed, and the balance is owned by Weyerhaeuser Company. The area lies largely within the timber-available area of the

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Ouachita National Forest. Within the past 20 years, only one small part of this watershed has been clearcut and replanted with genetically improved shortleaf pines, mostly in the eastern part of the watershed that was heavily damaged by a tornado in 1980. Other parts of the watershed have been thinning and burned by prescription. In this study, these treatments collectively represent a relatively low intensity of management.

North Alum Creek Watershed

The third watershed is North Alum Creek (9,800 acres), which lies north of the Alum Creek and Bread Creek watersheds in compartments 1445, 1446, 1447, and the north half of compartment 1456 of the Winona Ranger District. About half of this watershed is in national forest ownership, and Weyerhaeuser Company owns and manages the other half. Most of the industry land has been clearcut in the past 30 years and reforested with genetically improved loblolly pine. The majority of the national forest land lies within the timber-available land base and has been managed like the Bread Creek basin. For this study, the watershed represents a moderate intensity of forest management.

Glazypeau Watershed

The fourth watershed is the Glazypeau watershed (5,600 acres), approximately 4 miles southwest of Jessieville in Garland County, AR. Weyerhaeuser Company owns about 95 percent of this watershed, and the balance at the lower end of the drainage is in national forest. The area supported mixed shortleaf pine and pine-hardwood forests prior to the company's acquisition of the land in the late 1960s. Currently, a large portion of the Glazypeau watershed consists of genetically improved loblolly pine plantations established within the past 30 years. As such, it represents the most intensively managed watershed in the study.

Experimental Design

A key question in watershed research is how to quantify experimental error among watersheds. Experimental replication of watersheds of this size is impractical, because the variety of topographic, edaphic, and physiographic conditions in these watersheds invalidates the underlying assumption of homogeneity required to apply parametric statistical tests. Even if watersheds might have been identified that would have met this assumption, resources available to the project prohibit covering a larger area. As a result, alternatives to replication are needed to quantify treatment effects among watersheds.

Several different approaches to experimental design will be used in this study, depending on the particular resource variables and elements of interest. The first is to study baseline conditions in watersheds for several years prior to execution of the treatments, then to study conditions after treatment for a similar number of years. In this way, changes from baseline conditions can be compared across a number of years. This approach will generate valid statistical tests for some parameters, but will result only in descriptive information about other parameters. Yet given the size of the watersheds and the length of time over which measurements will occur, descriptive information will be valuable both for the scientific community and for land managers in the region.

A second approach is to subdivide watersheds into smaller units, and replicating those. Some elements of the research objectives will be met using small-scale studies within watersheds. For these, traditional parametric statistical analyses can be used. The elements for which this approach is feasible will vary among different research objectives.

A third approach is to use portions of the four watersheds to develop models of ecosystem attributes, and then to validate the models in the remaining parts of the watersheds. Work on model development will proceed by either subdivision of the existing watersheds or the establishment of separate watersheds for use in model validation. The use of locations outside the watershed boundaries that meet specific study objectives will be appropriate in some cases.

Finally, some of the results from the study will be published with the acknowledgment that data are descriptive rather than replicated; for some unique and less-intensively studied attributes or variables, descriptive data will be highly informative.

RESEARCH GROUPS AND EXPERIMENTAL APPROACHES

Wildlife Research

The wildlife research group has installed nearly 2,000 sample plots in 4 sets of 500 plots over 4 years. This sampling design serves as the pretreatment plot network for both the wildlife and the vegetation groups. The wildlife group measured habitat variables, and sampled populations of neotropical migrant and resident birds and herpetofauna.

Vegetation Research

The vegetation research group conducted annual pretreatment baseline surveys on the plot network established by the wildlife group. The resulting quantitative vegetation data base quantifies stand structure within and between watersheds. Vegetation measurements include overstory and midstory woody vegetation, as well as shrub and herbaceous vegetation. Because the vegetation and wildlife research measured the same plots, they may analyze data jointly if research dictates an interdisciplinary approach.

Hydrology Research

The hydrological network includes a series of nested, uncontrolled cross sections, three in each core watershed. These sampling stations gauge streamflow, which will allow scientists to quantify stage-discharge relationships. The study collects data on water quality as well as quantity in each watershed.

Aquatic Ecology Research

Aquatic ecological relationships are important in the overall landscape study design. Studies that quantify those relationships were established in streams within and between watersheds, as well as off the core watersheds. This research group studies the nature and composition of fish communities, the trophic relationships throughout the aquatic systems, and the changes in aquatic ecological relationships over time within and between watersheds.

Social Science Group

The social science group works primarily outside the core watersheds. Key responsibilities include studies on decision making and community involvement, advisory organizations, and forest users.

WATERSHED TREATMENTS

The study is designed to evaluate practical approaches to forest management on national forest lands. National forest and ranger district staff worked with research scientists to identify the desired future conditions proposed as treatments in this study. This process yielded more than four sets of desired future condition, so the three contiguous watersheds were subdivided for imposition of treatment conditions.

Baseline data collected prior to treatment versus data collected after treatment quantifies differences within and between watersheds as a result of treatment.

The unmanaged South Alum Creek watershed will support two desired future conditions. The first is an unmanaged control condition in the west half of the watershed, which will be maintained as an unharvested control block. In the east half of the watershed, a large contiguous block of single-tree selection will be imposed. Monitoring and research will be conducted to quantify the effects of this desired future condition across a large area in one operation.

The Bread Creek watershed will not be subdivided and will support continued management according to standard Forest Service practice. Reproduction cutting in this block will emphasize the seed-tree and group selection methods, and intermediate treatments will emphasize thinning. Prescribed fire in both immature and mature stands is a hallmark of standard operational practice on the Ouachita National Forest, and these stands will be subject to prescribed fire as called for in operational practice.

The North Alum Creek watershed will be subdivided into three subwatersheds that reflect different desired future conditions. In the first, reproduction cutting using the group selection method will be imposed using large groups between 2 and 10 acres, in which some groups will retain overstory trees and others will not. In the second, reproduction cutting under the group selection method will use small groups < 2 acres; again, some groups will have no residual trees, while others will. These group selection treatments are intended to create conditions appropriate for sampling wildlife species, especially birds, in openings of varied size and with different within-opening structures. The final treatment in a subunit of this large watershed is shortleaf pine-bluestem habitat restoration; this prescription reduces overstory density and midstory vegetation, and reintroduces prescribed fire to the landscape.

The Glazypeau watershed will not be subdivided. Standard industrial forest management practices will continue in this basin according to schedules and procedures of Weyerhaeuser Company. In this way, the effects of intensive forest management on a landscape will be quantified.

Scientists and managers will ask several questions as these treatments are imposed. For example, national forest management tactics shifted in the past decade from entries in individual compartments across a district to one larger scale entry in several contiguous compartments in a given year. The effect of concentrating management activities on terrestrial and aquatic ecosystems is not well documented. The phase III study can address elements of this question, such as quantifying the differences from baseline levels in each watershed. In many ways these treatments in the phase III landscape will represent the most intensive intervention within an agency project because they involve a concentrated action conducted across a large area in a short time frame.

On both national forest lands and forest industry lands in the phase III study, treatments will occur in an operational context. Within the national forest, treatments are planned according to standard agency practice. All reproduction cuttings and thinning will be prepared as commercial timber sales by the ranger district staff, subject to guidance by the research group only for meeting specific research objectives in residual stand condition. All timber sales were scheduled for the summer of 2000. Site preparation, prescribed burning, and treatment monitoring will commence when treatments have been completed.

SUMMARY

The phase III landscape-scale study was established to answer specific questions about forest management at scales larger than an individual stand. Because of difficulties with assumptions required to apply parametric statistical tests, the study uses several different sample designs. These include sampling over a number of years, discrete smaller studies of a more traditional statistical design, and subdivision of existing watersheds or identification of new watersheds for validation testing of models developed in the core watersheds. Treatments in the watersheds will be implemented as operational practice by national forest and industry land managers. Results from the phase III study will provide valuable guidance for land managers concerned about effects of forest practices at the landscape scale.

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LANDSCAPE CHARACTERIZATION OF FOUR WATERSHEDS UNDER DIFFERENT FOREST MANAGEMENT SCENARIOS IN THE OUACHITA MOUNTAINS OF ARKANSAS

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Abstract—Recent changes in philosophy concerning forest management have focused attention on managing ecosystems at scales beyond the stand level. Properties of forested landscapes, such as patch size and shape, edge density, and interspersion have direct influences on flora and fauna. However, there is little information regarding spatial patterns and processes across large-scale landscapes. We quantified landscape characteristics for four watersheds in the Ouachita Mountains that represent different ownerships and management objectives. The watersheds included (1) an intensively managed landscape predominantly under forest industry ownership, (2) a landscape that has historically been managed under USDA Forest Service standards and guidelines, (3) a landscape composed of a mixture of forest industry and USDA Forest Service lands, and (4) a landscape that has received minimal management and now is largely mature forest. Using multi-temporal Landsat Thematic Mapper satellite imagery in conjunction with field-collected data, landcover classifications were developed. This information was incorporated into a geographic information system and landscape metrics were computed. Metrics such as mean patch size and density, edge density, and interspersion varied by watershed. Differences in watersheds at both the watershed and landcover class levels suggest that the dynamics of some ecological interactions are likely to also be different in each watershed. However, the interpretation of specific watershed characteristics is dependant on the particular phenomena being investigated. The quantified differences in landscape characteristics of each watershed provide important information that can aid in making management decisions in our ecologically and sociologically complex forests.

INTRODUCTION

Recent philosophical changes in natural resource management have focused attention on managing ecosystems at scales beyond the stand level. This emphasis has led to a heightened conceptual and theoretical understanding of landscape patterns and functions. Landscape patterns are influenced by the composition and development of vegetation following disturbances, as well as the juxtaposition of these disturbances (Oliver 1981). However, there is a lack of information regarding these patterns across large-scale landscapes (Soulé and Kohm 1989, Jeffers 1988).

Properties of forested landscapes, such as patch size, edge density, and degree of fragmentation have a direct influence on flora and fauna (Hagan and others 1997, Rolstad 1991, MacArthur and Wilson 1967). Thus, increased knowledge of these interactions is needed to make informed management decisions regarding ecologically and sociologically complex landscapes. However, before studies can address questions involving landscape functions, landscape structure must be quantified. Landscape-scale research currently being conducted in the Ouachita Mountains of Arkansas is addressing a myriad of questions concerning landscape interactions of flora and fauna (Fox and others, in press; Guldin, in press; Tappe and others, in press). This paper compares structural landscape characteristics for four watersheds in which these interactions are being studied.

METHODS

Study Areas

We quantified landscape characteristics of four watersheds included in the Ouachita Mountain Ecosystem Management Research Project described by Guldin (in press). The watersheds were located in the Ouachita Mountains in Garland and Saline counties north of Hot Springs, Arkansas. Each represented different ownerships and management objectives as follows:

1. Little Glazypeau (LG)—2,275 ha predominantly under Weyerhaeuser Company ownership and intensively managed for wood products
2. North Alum Creek (NAC)—3,961 ha with approximately equal mixtures of Weyerhaeuser Company and USDA Forest Service ownership. Weyerhaeuser Company lands were intensively managed for wood products and USDA Forest Service lands were managed under multiple-use standards and guidelines
3. Bread Creek (BC)—1,535 ha predominantly under USDA Forest Service ownership and historically managed under multiple-use standards and guidelines
4. South Alum Creek (SAC)—1,499 ha predominantly under USDA Forest Service ownership and historically received minimal management.

Inherent in the differing proportions of ownership and management objectives of each watershed was an implicit

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continuum of intensity of forest management activity. The highest intensity of forest management activity occurred on LG, followed in descending order by NAC, BC, and SAC.

Remote Sensing and Landcover Classification

A Geographic Information System (GIS) was utilized to help characterize the study areas. Delineated forest stand boundaries on each watershed represented discrete administrative management units and not necessarily unique landcover types. Thus, spectral landcover classes initially were identified based on Landsat Thematic Mapper (TM) data (30-m resolution, bands 1–7) using scenes from February, April, and July, 1995. The 21 bands (7 from each time period) were clustered using an isodata clustering algorithm (unsupervised classification) with a maximum-likelihood classification to create 255 spectral classes (Richards and Jia 1999).

Transects with 50-m radius plots spaced approximately 200 m apart were established in each watershed. Vegetation data were collected during May–August 1995 on a total of 524 plots: 124 plots in LG, 235 plots in NAC, 75 plots in BC, and 90 plots in SAC. Similar to a classification scheme used by Hagan and others (1997), each plot was placed into 1 of 5 forest composition classes, 1 of 3 canopy cover classes, and 1 of 3 tree height classes (table 1). This information was then used to categorize the unsupervised 255 spectral classes into 12 landcover classes (table 2). This process used both the spectral information obtained from the unsupervised classification and ancillary GIS layers obtained from the Natural State Digital Database (http://sal.uamont.edu/sal/nsdd/) at the Spatial Analysis Laboratory, University of Arkansas - Monticello. Landcover information was incorporated into a GIS, and watershed-level and landcover class-level indices were computed using Fragstats v.2.0 (McGarigal and Marks 1994). These indices included percent of a landcover class comprising a watershed, patch density (PD), (number per 100 ha), largest patch index (LPI), (percent of a watershed comprised by the largest patch), mean patch size (MPS) and coefficient

Table 1—Plot characteristics used to categorize an unsupervised spectral classification into landcover classes

Characteristic	Category	Value range
Forest composition (%)	Pine	≥ 75 pine
	Pine/hardwood	≥ 60 pine < 75
	Mixed	≥ 40 pine or hardwood < 60
	Hardwood/pine	≥ 60 hardwood < 75
	Hardwood	≥ 75 hardwood
Canopy cover (%)	Low	0–33
	Medium	34–75
	High	76–100
Tree height (m)	Low	0–7
	Medium	> 7–13
	High	> 13

Table 2—Landcover classes used to characterize four watersheds under different forest management scenarios in the Ouachita Mountains of Arkansas

	Forest characteristics		
Landcover type	Composition	Canopy cover	Height
Young, open pine	Pine	Low	Low
Thin, open pine	Pine	Medium	Medium
Dense canopy pine	Pine	High	Medium
Sparse pine	Pine	Low	Medium
Pine/hardwood	Pine/hardwood	Medium	High
Mixed forest	Mixed	High	High
Hardwood/pine	Hardwood/pine	High	Medium
Hardwood	Hardwood	High	Medium
Sparse hardwood	Hardwood	Low	Medium
Grass	Grass	NA	NA
Rock/road	Rock/road	NA	NA
Water	Water	NA	NA

NA = not applicable.

of variation (CV), edge density (ED), (m per ha), mean shape index (MSI), (mean perimeter-to-area ratio), mean nearest neighbor distance (MNN), (mean distance from a patch to the nearest neighboring patch of the same type) and CV, and McGarigal and Mark's (1994) interspersion/juxtaposition index (IJI), (extent to which patch types are interspersed).

RESULTS

Area Metrics

The four watersheds differed in their landcover composition (table 3). Landcover classes composed of ≥ 75 percent pine comprised 50.3 percent of LG, 28.7 percent of NAC, 15.8 percent of BC, and 1.2 percent of SAC. Proportions of pine/hardwood were approximately 1.5 times greater in SAC than in BC, 2 times greater than in NAC, and 4 times greater than in LG. Proportions of mixed forest were approximately 1.1 times greater in SAC than in BC, 1.5 times greater than in NAC, and 3 times greater than in LG. Within other landcover classes, proportions varied between watersheds by < 8.5 percent (table 3).

Overall, LPI was highest for SAC, followed in decreasing order by BC, NAC, and LG (table 4). The LPI varied < 4.7 percent within all landcover classes with the exceptions of dense canopy pine and pine/hardwood classes (table 4). The largest patch of dense canopy pine comprised a larger proportion of LG than it did in any other watershed (approximately 6 times greater than NAC, 9 times greater than BC, and 36 times greater than SAC). The largest patch of pine/hardwood comprised a larger proportion of SAC than it did in any other watershed (approximately 1.5 times greater than BC, 8 times greater than NAC, and 12 times greater than LG). Young/open pine in BC had notably higher LPI than in other watersheds.

Patch Metrics

At the watershed-level, LG had approximately 19 patches per 100 ha more than NAC, 24 patches per 100 ha more

Table 3—Percent and area of 12 landcover classes within 4 watersheds under different forest management scenarios in the Ouachita Mountains of Arkansas

Landcover class	Little Glazypeau	North Alum Creek	Bread Creek	South Alum Creek
	- - - - - - - - - - -	- - - - - - *percent and area*	(*ha*) - - - - - -	- - - - - - - - - - -
Young/open pine	11.2 (255.4)	7.4 (292.3)	11.4 (174.6)	0.5 (7.2)
Thin/open pine	11.0 (249.3)	13.2 (524.3)	2.7 (41.5)	0.4 (6.4)
Dense canopy pine	24.4 (554.2)	7.9 (314.0)	1.7 (26.9)	0.3 (4.8)
Sparse pine	3.7 (84.6)	0.2 (7.4)	0.0 (0.0)	0.0 (0.0)
Pine/hardwood	11.3 (257.5)	24.5 (970.8)	35.6 (546.8)	46.6 (698.2)
Mixed forest	9.0 (204.6)	19.0 (751.1)	25.8 (395.2)	27.8 (416.5)
Hardwood/pine	12.5 (284.1)	16.2 (639.5)	10.7 (164.3)	7.89 (118.3)
Hardwood	9.5 (216.7)	8.8 (347.7)	10.1 (154.8)	14.1 (212.3)
Sparse hardwood	6.4 (145.0)	2.8 (111.7)	2.0 (30.5)	0.5 (7.6)
Grass	0.6 (14.4)	< 0.1 (1.6)	0.0 (0.0)	1.79 (26.7)
Rock/pavement	0.4 (8.4)	0.0 (0.0)	< 0.1 (0.1)	0.1 (1.2)
Water	< 0.1 (0.4)	< 0.01 (0.2)	0.0 (0.0)	0.0 (0.0)
Total	100.0 (2,274.8)	100.0 (3,960.6)	100.0 (15,34.7)	100.0 (1,499.2)

Table 4—Patch density and largest patch index for four watersheds under different forest management scenarios in the Ouachita Mountains of Arkansas

	Patch density and largest patch index			
Landcover class	Little Glazypeau	North Alum Creek	Bread Creek	South Alum Creek
	no./100 ha and (*percent*)			
Young/open pine	11.08	6.54	1.43	0.47
	(2.52)	(1.01)	(7.18)	(0.43)
Thin/open pine	14.29	6.34	6.45	0.27
	(1.33)	(4.39)	(0.48)	(0.25)
Dense canopy pine	4.26	4.67	1.82	0.67
	(7.16)	(1.17)	(0.76)	(0.20)
Sparse pine	1.63	0.03	—	—
	(1.83)	(0.19)	—	—
Pine/hardwood	9.32	9.85	8.21	9.07
	(3.31)	(4.91)	(22.02)	(39.34)
Mixed forest	14.29	15.07	14.6	17.01
	(0.63)	(1.03)	(3.96)	(2.53)
Hardwood/pine	16.66	17.24	19.68	16.61
	(1.07)	(1.31)	(0.49)	(0.56)
Hardwood	9.63	7.47	11.01	13.07
	(1.47)	(3.54)	(4.94)	(5.59)
Sparse hardwood	10.38	7.75	7.04	2.8
	(1.50)	(0.17)	(0.17)	(0.04)
Grass	1.71	0.3	—	0.47
	(0.13)	(0.01)	—	(1.70)
Rock/pavement	0.84	—	0.07	0.07
	(0.12)	—	(0.01)	(0.08)
Water	0.09	0.03	—	—
	(0.01)	(< 0.01)	—	—
All classes	94.18	75.28	70.31	60.49
	(7.16)	(4.91)	(22.02)	(39.34)

than BC, and 33 patches per 100 ha more than BC (table 4). PD for young per open pine was highest in LG, followed in descending order by NAC, BC, and SAC. PD was also highest in LG and lowest in SAC for thin/open pine, with NAC and BC having similar PD. Within other landcover classes, PD varied among watersheds by < 8 patches per 100 ha.

Overall, SAC had a MPS 0.23 ha larger than BC, 0.32 ha larger than NAC, and 0.59 ha larger than LG (table 5). MPS followed a similar pattern for the pine/hardwood class. MPS for dense canopy pine followed a reverse pattern, with LG having the largest MPS, followed in decreasing order by NAC, BC, and SAC. Young/open pine in BC had notably higher MPS than in other watersheds.

Overall, relative variability of MPS was greatest for SAC, with MPS CV approximately 1.5 times greater than that for BC, and 2 times greater than that for NAC and LG (table 5). MPS CV followed a similar pattern for the mixed forest class. MPS CV for young/open and dense canopy pine in LG, followed a reverse pattern, with LG having the largest MPS CV, followed in decreasing order by NAC, BC, and SAC.

Edge and Shape Metrics

At the watershed-level, LG had approximately 11 m per ha more edge than NAC, 19 m per ha more edge than BC, and 34 m per ha more edge than SAC (table 6). ED followed a similar pattern for all landcover classes composed of ≥ 75 percent pine, with LG having, on average, approximately 19 m per ha more edge than NAC, 41 m per ha more edge than BC, and 56 m per ha more edge than SAC. Pine/hardwood and mixed forest classes followed a reverse pattern. ED of the pine/hardwood class in SAC was approximately 21 m per ha more than in BC, 50 m per ha more than in SAC, and 77 m per ha more than in LG. ED of the mixed pine forest class in SAC was approximately 4 m per ha more than in BC, 35 m per ha more than in SAC, and 80 m per ha more than in LG.

Table 5—Mean patch size and coefficient of variation for four watersheds under different forest management scenarios in the Ouachita Mountains of Arkansas

Landcover class	Little Glazypeau	North Alum Creek	Bread Creek	South Alum Creek
	ha and percent			
Young/open pine	1.01 (525.43)	1.13 (381.24)	7.94 (292.19)	1.03 (216.41)
Thin/open pine	0.77 (371.89)	2.09 (585.17)	0.42 (207.58)	1.6 (89.82)
Dense canopy pine	5.71 (389.99)	1.70 (285.52)	0.96 (240.71)	0.49 (180.53)
Sparse pine	2.29 (314.38)	7.38 (0.00)	— —	— —
Pine/hardwood	1.21 (462.00)	2.49 (523.14)	4.34 (701.76)	5.13 (980.97)
Mixed forest	0.63 (215.20)	1.26 (263.06)	1.76 (298.34)	1.63 (270.75)
Hardwood/pine	0.75 (304.52)	0.94 (339.43)	0.54 (194.65)	0.47 (213.04)
Hardwood	0.99 (341.98)	1.17 (721.50)	0.92 (641.86)	1.08 (562.32)
Sparse hardwood	0.61 (491.84)	0.36 (189.24)	0.28 (151.95)	0.18 (77.37)
Grass	0.37 (147.58)	0.14 (79.35)	— —	3.83 (231.59)
Rock/pavement	0.44 (145.21)	— —	0.09 (0.00)	1.17 (0.00)
Water	0.18 (< 0.01)	0.18 (0.00)	— —	— —
All classes	1.06 (561.47)	1.33 (531.84)	1.42 (811.96)	1.65 (1,205.51)

Overall, MSI was similar for all watersheds (table 6). No consistent trends in MSI were observed at the landcover class level.

Nearest Neighbor and Interspersion Metrics

Overall, MNN was similar for all watersheds (table 7). Likewise, no consistent trends in MNN were observed at the landcover class level. However, SAC had a notably higher MNN than other watersheds for both the thin/open (averaging approximately 675 m more) and dense canopy pine (averaging approximately 227 m more) classes.

Overall, relative variability of MNN was greatest for LG, followed by SAC (table 7). MNN CV for LG and SAC were approximately 1.5 to 2 times greater than those for NAC and BC. No consistent trends in MNN CV were observed at the landcover class level. However, SAC had a notably higher MNN CV for both the hardwood pine and hardwood classes, each averaging approximately 1.5 times greater than in other watersheds. The MNN CV for the young/open pine class in BC also had a notable CV, averaging 1.5 times greater than in other watersheds.

At the watershed level, the interspersion of available patch types is about 75 percent of the maximum possible equitable distribution in LG, whereas NAC and BC are at about 65 percent and SAC is below 50 percent (table 8). The IJI is < 50 percent for the pine/hardwood and mixed forest classes in SAC, and > 80 percent for the young/open pine class in BC, the thin/open pine class in SAC, and the hardwood/pine class in LG.

CONCLUSIONS

Several landscape characteristics suggest a gradient of spatial heterogeneity represented by differing proportions of ownership, management objectives, and intensity of forest management activity. At the watershed level, PD, ED, and IJI all were highest for LG, followed in descending order by

Table 6—Edge density and mean shape index for four watersheds under different forest management scenarios in the Ouachita Mountains of Arkansas

	Edge density and mean shape index			
Vegetation class	Little Glazypeau	North Alum Creek	Bread Creek	South Alum Creek
	m/ha and (*percent*)			
Young/open pine	47.83 (1.28)	30.25 (1.26)	24.65 (1.51)	1.88 (1.19)
Thin/open pine	68.52 (1.36)	48.22 (1.45)	20.00 (1.23)	2.12 (1.55)
Dense canopy pine	59.94 (1.48)	31.66 (1.38)	8.02 (1.26)	1.80 (1.22)
Sparse pine	10.99 (1.32)	0.53 (1.93)	— —	— —
Pine/hardwood	54.44 (1.35)	81.21 (1.38)	110.37 (1.47)	131.28 (1.38)
Mixed forest	62.06 (1.34)	106.50 (1.49)	137.10 (1.58)	141.57 (1.50)
Hardwood/pine	80.07 (1.38)	94.89 (1.38)	76.59 (1.32)	55.24 (1.24)
Hardwood	42.87 (1.30)	31.09 (1.22)	37.59 (1.22)	59.34 (1.27)
Sparse hardwood	36.51 (1.23)	22.49 (1.22)	17.20 (1.16)	5.32 (1.11)
Grass	5.03 (1.24)	0.48 (1.07)	— —	3.44 (1.26)
Rock/pavement	2.31 (1.16)	— —	0.08 (1.00)	0.50 (1.66)
Water	0.16 (1.06)	0.05 (1.06)	— —	— —
All classes	235.36 (1.33)	223.68 (1.36)	215.80 (1.35)	201.25 (1.34)

NAC, BC, and SAC. Conversely, LPI and MPS were largest for SAC, followed in descending order by BC, NAC, and LG. These observed landscape characteristics suggest that the watersheds can be ranked in order of decreasing spatial heterogeneity as LG > NAC > BC > SAC.

For most landcover classes composed of ≥ 75 percent pine, larger, more dense patches were well interspersed in LG than in the other watersheds. Additionally, they comprised 50 percent of the area of LG. In contrast, larger pine/hardwood patches occurred in SAC, with the largest accounting for 39 percent of the watershed area. Mixed forest classes comprised > 25 percent of BC and SAC, where they also had their largest MPS and ED. However, interspersion was poor (< 50 percent) in SAC. Similar proportions and patch sizes of the hardwood/pine class occurred in all watersheds. The hardwood class comprised < 15 percent of all watersheds, but proportionally occurred most in SAC.

LG was characterized by a higher proportion of pine landcover classes, more and smaller patches, more edge, and higher interspersion than the other watersheds. SAC was characterized by a higher proportion of pine-hardwood and mixed forest landcover classes, fewer and larger patches, less edge, and lower interspersion than the other watersheds. The NAC and BC watersheds were intermediate in these measures and varied relative to the intensity of forest management. The differences in watersheds at both the watershed and class levels indicate that the dynamics of some ecological interactions are likely to also be different in each watershed. However, the interpretation of specific watershed characteristics is dependant on the particular phenomena being investigated.

Table 7—Mean nearest neighbor and coefficient of variation for four watersheds under different forest management scenarios in the Ouachita Mountains of Arkansas

	Mean nearest neighbor and coefficient of variation			
Landcover class	Little Glazypeau	North Alum Creek	Bread Creek	South Alum Creek
	m and (*percent*)			
Young/open pine	54.19 (77.49)	56.43 (76.22)	145.66 (113.87)	124.28 (72.33)
Thin/open pine	51.58 (73.43)	43.72 (63.62)	53.03 (61.16)	724.75 (89.97)
Dense canopy pine	54.34 (130.96)	53.99 (89.29)	98.94 (206.01)	305.45 (158.71)
Sparse pine	56.99 (106.15)	0.00 (0.00)	— —	— —
Pine/hardwood	62.14 (91.00)	46.75 (73.38)	41.16 (48.07)	41.20 (60.35)
Mixed forest	51.39 (87.56)	42.16 (76.77)	37.80 (71.43)	37.89 (47.87)
Hardwood/pine	45.66 (75.57)	46.04 (73.80)	50.52 (64.14)	65.84 (102.00)
Hardwood	67.59 (103.92)	83.72 (92.58)	72.62 (88.68)	60.57 (154.24)
Sparse hardwood	71.85 (84.18)	77.57 (96.95)	90.37 (72.56)	79.70 (109.69)
Grass	163.86 (122.83)	332.91 (111.72)	— —	206.77 (81.24)
Rock/pavement	190.53 (109.78)	— —	0.00 (0.00)	0.00 (0.00)
Water	4,185.62 (< 0.01)	0.00 (0.00)	— —	— —
All classes	63.10 (223.63)	54.70 (104.24)	57.70 (106.67)	60.90 (175.79)

Table 8—Interspersion/juxtaposition index for four watersheds under different forest management scenarios in the Ouachita Mountains of Arkansas

	Interspersion/juxtaposition index			
Landcover class	Little Glazypeau	North Alum Creek	Bread Creek	South Alum Creek
	percent			
Young/open pine	71.80	67.50	80.85	77.06
Thin/open pine	62.04	65.42	68.14	84.10
Dense canopy pine	56.45	52.76	64.39	68.12
Sparse pine	53.41	26.94	—	—
Pine/hardwood	67.77	57.36	49.46	44.32
Mixed forest	69.55	58.40	58.15	46.51
Hardwood/pine	81.85	66.61	62.39	60.09
Hardwood	74.01	61.88	73.42	50.95
Sparse hardwood	78.19	66.88	73.08	60.65
Grass	78.71	53.57	—	66.22
Rock/pavement	72.72	—	33.33	53.40
Water	44.92	19.57	—	—
All classes	75.26	66.53	64.37	48.78

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BREEDING BIRD COMMUNITIES ON FOUR WATERSHEDS UNDER DIFFERENT FOREST MANAGEMENT SCENARIOS IN THE OUACHITA MOUNTAINS OF ARKANSAS

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Abstract—Concern for many migratory landbird species has led to an increased emphasis on managing ecosystems at scales beyond the stand level. We characterized breeding bird numbers, species richness, diversity, and evenness on four watersheds in the Ouachita Mountains of west-central Arkansas. These four watersheds represented different ownerships and management objectives, and thus differed markedly in many respects, including landcover composition and distribution, mean patch size, and edge density. Point count surveys were conducted by three observers on approximately 525 different 50-m radius plots each spring during 1995-1998, for a total of 2,108 plots. A total of 19,030 birds were encountered representing 97 species. Patterns in numbers of individuals, species richness, and diversity appeared associated with the level of spatial heterogeneity and landscape composition in each watershed. Numbers of individuals per plot, species per plot, and diversity per plot for migrant and resident species were generally all highest in watersheds with high levels of management activity and lowest in watersheds with low levels of management activity. Watersheds were similar in regard to cavity and canopy nesting bird community characteristics. Ground and shrub nesting bird community characteristics were related to amounts of available early successional habitats. Species of special concern were represented in all watersheds.

INTRODUCTION

Concern for many migratory landbird species has led to an increased emphasis on managing ecosystems at scales beyond the stand level. Birds are often viewed as indicators of ecosystem health due to their sensitivity to environmental variability (Maurer 1993). Growing evidence suggests that populations of many species of neotropical migratory birds (NTMB), (species that migrate each year between temperate breeding areas and tropical wintering areas) are declining and that these declines have accelerated in recent years (Robbins and others, 1989; Askins and others, 1990; Finch 1991). Available information suggests that these populations may be limited by circumstances on both their wintering and breeding habitat, as well as along their migration routes (Sherry and Holmes 1993). Factors that have been implicated on their breeding range include habitat destruction/alteration, forest fragmentation, increased nest parasitism by brown-headed cowbirds (*Molothrus ater*), and increased predation, especially in edge-dominated habitat.

Substantial knowledge exists on stand-level avian community relationships, but information is needed to assess the impacts of different types, intensities, and spatial arrangements of forest management practices on NTMB at the landscape/watershed level. In the Ozark-Ouachita Highland physiographic area, Hunter and others (1993) identified several neotropical migratory species to be declining in numbers, including 11 of 22 species that inhabit mature forest and 4 of 6 species that inhabit forest edges. Where landscapes are fragmented by agriculture/urbanization, fragmentation has been implicated in NTMB declines. However, the impacts on bird populations of forest fragmentation that result in a mosaic of different stand compositions and ages with little or no agriculture/urban inclusions (characteristic of much of the Ouachita Mountains) are unknown.

This paper reports on results of a 4-year study (1995 – 1998) of breeding bird communities on four watersheds under different levels of forest management intensity. Our objectives were to (1) compare avian community and nesting guild characteristics (numbers of individuals, species richness, diversity, and evenness) across watersheds, and (2) relate these community characteristics to landscape attributes. We documented community characteristics of migrant and resident species, as well as three nesting guilds: canopy nesters, cavity nesters, and ground and shrub nesters.

METHODS

Study areas

Four watersheds included in the Ouachita Mountain Ecosystem Management Research Project described by Guldin (this volume) and Tappe and others (this volume) served as the study areas. The watersheds were located in Garland and Saline Counties in the Ouachita Mountains near Hot Springs, Arkansas. Past and current forest management practices on each watershed reflected different combinations of forest ownership and distinct management strategies, thus providing a range of intensity of management activities. Consequently, these watersheds differ markedly in many respects, including landcover composition and distribution characteristics, mean patch size, and edge density (Tappe and others, in press).

The Little Glazypeau watershed (LG), consisting of 2,275 ha and managed largely for sawlog production by Weyerhaeuser

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Company, represented the most intensively managed watershed. Much of the second-growth shortleaf pine (*Pinus echinata*)-hardwood forest that originally covered this watershed had been harvested and planted to loblolly pine (*P. taeda*) plantations that ranged in size from 9 to 142 ha. These plantations are normally thinned twice, pruned up to approximately 5 to 8 m high, fertilized, and harvested at 30 to 35 years of age. Other unharvested and selectively harvested acreage in the watershed occurs on rocky ridge tops, on steep slopes, and in streamside management zones that were retained for watershed protection and to provide wildlife habitat diversity.

The North Alum Creek watershed (NAC) was of mixed ownership, with about half of the area under Weyerhaeuser Company management and half under USDA Forest Service management. This watershed encompasses 3,961 ha of land under a diverse range of management activities, ranging from no management to intensively managed loblolly pine plantations similar to that in the LG watershed. USDA Forest Service land within this watershed is primarily mixed shortleaf pine–hardwood forest and has been managed using a mixture of silvicultural approaches to obtain multiple objectives as set by the Ouachita National Forest.

The 1,535-ha Bread Creek watershed (BC), primarily USDA Forest Service land, has been managed according to prevailing Forest Service standards and guides for several decades. It is composed primarily of mixed shortleaf pine–hardwood forest that has been managed similar to USDA Forest Service land located in the NAC watershed.

The South Alum Creek watershed (SAC) represented the lowest management intensity. This 1,499-ha watershed, which is designated as the Alum Creek Experimental Forest, is almost entirely USDA Forest Service land and has received minimal management over the past several decades; it now consists of a mature, mixed forest over the majority of the area.

More detailed information on landcover characteristics and patch and landscape attributes can be found in Tappe and others (in press). Tappe and others (in press) ranked the watersheds in order of decreasing spatial and compositional heterogeneity as LG>NAC>BC>SAC.

Bird Surveys

Because of the large sizes of the watersheds, adequate coverage of all portions of each watershed was not logistically possible within a single year using permanent points. Thus, we installed 505 – 550 plots each year to allow for sampling all portions of each watershed. In 1995, nearly 113 km of transects with plots located approximately 200 m apart were established over the four watersheds. In 1996, the same transects were used, but new plots were located between the 1995 plots by shifting plot centers 100 m. In 1997, approximately 113 km of new transects were established between the original transects and a new set of plots were used. In 1998, new plots were located between the 1997 plots, along the same transects. This resulted in 2,108 sample plots over the duration of the study (table 1).

Birds were sampled between May 6 and June 9 each year using 50-m fixed-radius point counts. Each plot was censused once by three different observers. All counts lasted 5 minutes and were conducted within 3.5 hours of sunrise on days with little or no rain and with winds < 6-11 kph.

Analyses

Bird counts from each observer were combined to provide one data set for each point sampled within a given year. Species that are not known to breed in the Ouachita Mountains of Arkansas were not used in any analyses. Numbers of individuals, species richness, diversity (Shannon and Weiner 1963), and evenness were computed for each point for all breeding birds combined and for each of the following subsets: migrants (short- and long-distance combined), residents, canopy nesters, cavity nesters, and ground and shrub nesters. A 2-way analysis-of-variance was used to test ($\alpha \leq 0.05$) for year, watershed, and year-watershed interaction effects. If a year-watershed interaction existed, differences in the watershed effect were interpreted within years using a one-way analysis-of-variance. Mean separation was accomplished using Fisher's least significant differences (LSD) procedure. Because watersheds could not be replicated, means across the three observers within years constituted pseudoreplication. This limited our inference space, and thus constrained our results and conclusions to the four specific watersheds on which the study was conducted.

Table 1—Number of points used each year for sampling breeding bird communities on four watersheds under different levels of forest management activity in the Ouachita Mountains of Arkansas

	Year				
Watershed	1995	1996	1997	1998	Total
	- - - - - - - - - - - - - - - - - - *number* - - - - - - - - - - - - - - - - - -				
Little Glazypeau	124	117	154	141	536
North Alum Creek	235	226	232	224	917
Bread Creek	75	75	76	76	302
South Alum Creek	90	87	88	88	353
Total	524	505	550	529	2,108

Table 2—Species list[a] and frequencies of occurrence of birds identified during 50-m fixed-radius point counts during May–June 1995 through 1998 on four watersheds under different levels of forest management activity in the Ouachita Mountains of Arkansas[b]

				Birds per 100 plots			
Species	Scientific name	*n*	Total	LG	NAC	BC	SAC
			percent				
Red-eyed vireo	*Vireo olivaceus*	3,219	16.92	126.62	173.77	144.92	141.41
Pine warbler	*Dendroica pinus*	1,965	10.33	79.48	83.72	102.95	128.17
Indigo bunting	*Passerina cyanea*	1,556	8.18	107.02	81.64	45.90	25.35
Black and white warbler	*Mniotilta varia*	1,150	6.04	63.96	63.06	42.62	27.32
Carolina chickadee	*Parus carolinensis*	858	4.51	56.38	36.28	35.41	31.83
White-eyed vireo	*Vireo griseus*	798	4.19	75.60	37.38	14.43	0.85
Blue-gray gnatcatcher	*Polioptila caerulea*	708	3.72	75.42	23.83	21.64	4.51
Ovenbird	*Seiurus aurocapillus*	659	3.46	4.81	32.13	31.48	68.45
Yellow-breasted chat	*Icteria virens*	635	3.34	61.92	27.21	16.07	0.56
Prairie warbler	*Dendroica discolor*	613	3.22	55.64	26.34	22.62	0.56
Scarlet tanager	*Piranga olivacea*	592	3.11	19.96	31.26	25.57	33.80
Common yellowthroat	*Geothlypis trichas*	484	2.54	51.39	20.87	4.59	0.28
Tufted-titmouse	*Parus bicolor*	460	2.42	27.17	22.30	11.48	20.85
Hooded warbler	*Wilsonia critrina*	392	2.06	37.15	19.45	3.28	0.85
Carolina wren	*Thryothorus ludovicianus*	371	1.95	34.94	14.21	7.87	7.89
Cedar waxwing[b]	*Bombycilla cedrorum*	337	1.77	18.11	17.38	16.39	8.45
Northern cardinal	*Cardinalis cardinalis*	296	1.56	32.72	9.84	9.18	0.28
Kentucky warbler	*Oporomis formosus*	289	1.52	29.39	11.80	6.89	0.28
Summer tanager	*Piranga rubra*	272	1.43	16.45	13.11	7.21	11.55
Pileated woodpecker	*Dryocopus pileatus*	247	1.30	8.32	11.37	11.48	17.75
Great-crested flycatcher	*Myiarchus crinitus*	233	1.22	11.65	11.80	7.54	10.99
Yellow-billed cuckoo	*Coccyzus americanus*	210	1.10	7.21	13.44	10.16	4.79
American crow	*Corvus brachyrhynchos*	207	1.09	12.38	9.51	6.23	9.58
Bluejay	*Cyanocitta cristata*	200	1.05	5.36	7.43	12.79	18.03
American goldfinch	*Carduelis tristis*	194	1.02	23.84	4.81	2.30	3.94
Worm-eating warbler	*Helmitheros vermivorus*	189	0.99	11.46	10.93	5.57	2.82
Field sparrow	*Spizella pusilla*	188	0.99	19.22	7.32	5.25	0.28
Acadian flycatcher	*Empidonax virescens*	165	0.87	11.46	9.62	1.64	2.82
Rufous-sided towhee	*Pipilo erythrophthalmus*	132	0.69	14.42	5.68	0.66	0.00
Brown-headed cowbird	*Molothrus ater*	129	0.68	21.26	1.31	0.33	0.28
Hairy woodpecker	*Picoides villosus*	123	0.65	2.40	6.56	7.87	7.32
White-breasted nuthatch	*Sitta carolinensis*	87	0.46	2.03	6.23	0.98	4.51
Downy woodpecker	*Picoides pubescens*	76	0.40	7.58	2.62	2.95	0.56
Northern bobwhite	*Colinus virginianus*	73	0.38	4.81	4.48	1.64	0.28
Swainson's thrush[b]	*Catharus ustulatus*	68	0.36	1.85	4.59	4.59	0.56
Wild turkey	*Meleagris gallopavo*	66	0.35	1.66	4.15	1.97	3.66
Yellow-throated vireo	*Vireo flavifrons*	64	0.34	6.28	2.73	0.98	0.56
Eastern wood pewee	*Contopus virens*	63	0.33	7.21	1.97	1.31	0.56
Louisiana waterthrush	*Seiurus motacilla*	57	0.30	3.33	2.51	1.97	2.82
Broad-winged hawk	*Buteo platypterus*	52	0.27	2.03	2.51	1.97	3.38
Ruby-throated hummingbird	*Archilochus colubris*	46	0.24	4.62	2.08	0.00	0.56
Tennessee warbler[b]	*Vermivora peregrina*	37	0.19	1.85	2.19	1.31	0.85
Chimney swift	*Chaetura pelagica*	34	0.18	0.37	1.20	2.62	3.66
Northern parula warbler	*Parula americana*	29	0.15	4.81	0.22	0.33	0.00
Rose-breasted grosbeak[b]	*Pheucticus ludovicianus*	26	0.14	0.37	1.64	0.66	1.97
Black-throat green warbler[b]	*Dendroica townsendi*	24	0.13	2.40	0.44	0.66	1.41
Common flicker	*Colaptes auratus*	21	0.11	1.11	0.87	0.98	1.13
Blue grosbeak	*Guiraca caerulea*	21	0.11	0.92	1.09	1.97	0.00
Chuck-wills widow	*Caprimulgus carolinensis*	18	0.09	1.48	0.87	0.00	0.56
Mourning dove	*Zenaida macroura*	18	0.09	2.96	0.00	0.33	0.28
Wood thrush	*Hylocichla mustelina*	17	0.09	0.55	1.09	0.66	0.56
Red-shouldered hawk	*Buteo lineatus*	17	0.09	1.11	0.11	0.66	2.25
Brown thrasher	*Toxostoma rufum*	17	0.09	1.11	0.66	0.98	0.56

continued

Table 2—Species list[a] and frequencies of occurrence of birds identified during 50-m fixed-radius point counts during May–June 1995 through 1998 on four watersheds under different levels of forest management activity in the Ouachita Mountains of Arkansas[b] (continued)

				Birds per 100 plots			
Species	Scientific name	*n*	Total	LG	NAC	BC	SAC
			percent				
American redstart	*Setophaga ruticilla*	17	0.09	1.29	1.09	0.00	0.00
Bay-breasted warbler[b]	*Dendroica castarea*	15	0.08	2.40	0.22	0.00	0.00
Magnolia warbler[b]	*D. magnolia*	15	0.08	0.55	0.77	1.31	0.28
Gray catbird	*Dumetella carolinensis*	15	0.08	0.92	0.87	0.66	0.00
Baltimore oriole	*Icterus galbula*	14	0.07	0.37	1.20	0.00	0.28
Red-bellied woodpecker	*Melanerpes carolinus*	14	0.07	1.85	0.44	0.00	0.00
Turkey vulture	*Cathartes aura*	9	0.05	0.18	0.87	0.00	0.00
Chipping sparrow	*Spizella paserina*	9	0.05	0.55	0.66	0.00	0.00
Veery[b]	*Catharus fuscescens*	8	0.04	0.37	0.33	0.00	0.85
Red-tailed hawk	*Buteo jamaicensis*	7	0.04	0.92	0.11	0.33	0.00
Chestnut-sided warbler[b]	*Dendroica pensylvanica*	7	0.04	1.11	0.11	0.00	0.00
Barred owl	*Strix varia*	7	0.04	0.18	0.33	0.00	0.85
Sharp-shinned hawk	*Accipiter striatus*	6	0.03	0.18	0.11	0.33	0.85
Orchard oriole	*Icterus spurius*	6	0.03	1.11	0.00	0.00	0.00
Belted kingfisher	*Ceryle alcyon*	6	0.03	0.18	0.33	0.33	0.28
Whip-poor-will	*Caprimulgus vociferus*	6	0.03	0.74	0.11	0.00	0.28
Nashville warbler[b]	*Vermivora ruficapilla*	6	0.03	0.18	0.33	0.66	0.00
Eastern kingbird	*Tyranus tyranus*	6	0.03	1.11	0.00	0.00	0.00
Eastern Phoebe	*Sayomis phoebe*	5	0.03	0.37	0.22	0.00	0.28
Swainson's warbler	*Limnothlypis swainsonii*	5	0.03	0.37	0.33	0.00	0.00
Eastern bluebird	*Sialia sialis*	5	0.03	0.74	0.11	0.00	0.00
Yellow warbler[b]	*Dendroica petechia*	4	0.02	0.18	0.33	0.00	0.00
Cooper's hawk	*Accipiter cooperii*	3	0.02	0.55	0.00	0.00	0.00
Brown-headed nuthatch	*Sitta pusilla*	3	0.02	0.00	0.33	0.00	0.00
Purple martin	*Progne subis*	3	0.02	0.00	0.33	0.00	0.00
Warbling vireo	*Vireo gilvus*	3	0.02	0.00	0.33	0.00	0.00
Great blue heron	*Ardea herodias*	2	0.01	0.00	0.22	0.00	0.00
Hermit thrush[b]	*Catharas guttatus*	2	0.01	0.00	0.22	0.00	0.00
White-throated sparrow[b]	*Zonotrichia albicollis*	2	0.01	0.37	0.00	0.00	0.00
Golden-crowned kinglet[b]	*Regulus satrapa*	2	0.01	0.18	0.00	0.00	0.28
Wilson warbler[b]	*Wilsonia pusilla*	2	0.01	0.18	0.00	0.33	0.00
Red-breasted nuthatch[b]	*Sitta canadensis*	2	0.01	0.37	0.00	0.00	0.00
Cerulean warbler	*Dendroica cerulea*	1	0.01	0.00	0.11	0.00	0.00
Yellow-bellied flycatcher[b]	*Empidonax flaviventris*	1	0.01	0.00	0.00	0.00	0.28
Blue-winged warbler[b]	*Vermivora pinus*	1	0.00	1.00	0.00	0.00	0.28
Red crossbill[b]	*Loxia curvirostra*	1	0.01	0.00	0.00	0.00	0.28
Red-winged blackbird	*Agelaius phoeniceus*	1	0.01	0.00	0.11	0.00	0.00
Great horned owl	*Bubo virginianus*	1	0.01	0.00	0.00	0.33	0.00
Song sparrow[b]	*Melospiza melodia*	1	0.01	0.00	0.00	0.33	0.00
Wood duck	*Aix sponsa*	1	0.01	0.00	0.11	0.00	0.00
Screech owl	*Otus asio*	1	0.01	0.00	0.11	0.00	0.00
Canada warbler[b]	*Wilsonia canadensis*	1	0.01	0.18	0.00	0.00	0.00
American woodcock	*Scolopax minor*	1	0.01	0.18	0.00	0.00	0.00
Blackburnian warbler[b]	*Dendroica fusca*	1	0.01	0.00	0.00	0.33	0.00
Total		19,030	100.00				

LG = Little Glazypeau; NAC = North Alum Creek; BC = Bread Creek; SAC = South Alum Creek.

[a] Common and scientific names follow Hamel (1992).

[b] Nonbreeding species.

RESULTS

A total of 19,030 birds representing 97 species were recorded over all watersheds and years (table 2). Six species comprised approximately 50 percent of the individuals recorded: red-eyed vireo (*Vireo olivaceus*), pine warbler (*Dendroica pinus*), indigo bunting (*Passerina cyanea*), black and white warbler (*Mniotilta varia*), Carolina chickadee (*Parus carolinensis*), and white-eyed vireo (*Vireo griseus*). With the exception of the rufous-sided towhee (*Pipilo erythrophthalmus*), species for which >50 individuals were recorded occurred in all watersheds. A large portion of species occurred most frequently in LG, followed in decreasing order by NAC, BC, and SAC. The pine warbler, ovenbird (*Seiurus aurocapillus*), and pileated woodpecker (*Dryocopus pileatus*) followed an opposite pattern, with their highest numbers occurring in SAC. Species such as the red-eyed vireo and the tufted-titmouse (*Parus bicolor*) occurred in similar numbers across all watersheds. Recorded numbers of nonbreeding migrants (table 2) were a reflection of the timing of our bird surveys and not a result of habitat differences between watersheds.

Of the 97 recorded species, 76 are known to breed in the Ouachita Mountains of Arkansas. Of those breeding birds, year-watershed interactions existed for numbers of individuals per plot ($F = 3.63$; df = 9, 2092; $P < 0.001$), species per plot ($F = 3.76$; df = 9, 2092; $P < 0.001$), and diversity per plot ($F = 3.10$; df = 9, 2092; $P = 0.001$). Thus, these

Table 3—Mean (SE) breeding bird community metrics recorded on four watersheds under different levels of forest management activity in the Ouachita Mountains of Arkansas

	Year			
Watershed	1995	1996	1997	1998
	number of individuals per plot			
Little Glazypeau	8.81 A[a]	13.50 A	13.95 A	11.91 A
	(0.452)	(0.654)	(0.484)	(0.485)
North Alum Creek	6.02 B	8.33 B	11.36 B	10.30 B
	(0.266)	(0.289)	(0.354)	(0.422)
Bread Creek	4.61 C	5.88 C	8.73 C	8.07 C
	(0.359)	(0.357)	(0.652)	(0.533)
South Alum Creek	4.02 C	5.67 C	9.26 C	6.34 C
	(0.280)	(0.316)	(0.413)	(0.429)
	$F = 30.44$	$F = 59.88$	$F = 22.20$	$F = 20.77$
	df = 3, 520	df = 3, 501	df = 3, 546	df = 3, 525
	$P < 0.001$	$P < 0.001$	$P < 0.001$	$P < 0.001$
	number of species per plot			
Little Glazypeau	5.56 A	7.77 A	9.03 A	6.87 A
	(0.240)	(0.290)	(0.250)	(0.240)
North Alum Creek	3.90 B	5.13 B	6.96 B	5.86 B
	(0.140)	(0.170)	(0.200)	(0.200)
Bread Creek	3.25 C	3.81 C	5.05 C	4.88 C
	(0.230)	(0.230)	(0.330)	(0.270)
South Alum Creek	2.63 C	3.54 C	5.47 C	3.74 D
	(0.150)	(0.170)	(0.210)	(0.200)
	$F = 35.60$	$F = 63.00$	$F = 45.64$	$F = 20.56$
	df = 3, 520	df = 3, 501	df = 3, 546	df = 3, 525
	$P < 0.001$	$P < 0.001$	$P < 0.001$	$P < 0.001$
	diversity per plot			
Little Glazypeau	1.50 A	1.83 A	2.02 A	1.72 A
	(0.049)	(0.041)	(0.032)	(0.037)
North Alum Creek	1.13 B	1.41 B	1.73 B	1.51 B
	(0.039)	(0.035)	(0.029)	(0.036)
Bread Creek	0.96 C	1.13 C	1.35 C	1.38 B
	(0.069)	(0.061)	(0.068)	(0.053)
South Alum Creek	0.80 C	1.08 C	1.52 D	1.10 C
	(0.054)	(0.052)	(0.044)	(0.057)
	$F = 29.59$	$F = 48.53$	$F = 45.48$	$F = 28.19$
	df = 3, 520	df = 3, 501	df = 3, 546	df = 3, 525
	$P < 0.001$	$P < 0.001$	$P < 0.001$	$P < 0.001$

[a] Means in a column followed by the same letter did not differ ($P > 0.05$).

variables were interpreted within years. Numbers of individuals per plot, species per plot, and diversity per plot were consistently highest in LG each of the 4 years, followed by NAC (table 3). BC and SAC recorded similar numbers each year, with SAC having the lowest species per plot in 1998 and the lowest diversity per plot in 1997 and 1998. No year-watershed interaction was found for evenness per plot (F = 0.95; df = 9, 1985; P = 0.483), and evenness per plot did not differ (F = 2.89; df = 3, 9; P = 0.094) among watersheds (LG = 0.94, SE = 0.002; NAC = 0.94, SE = 0.002; BC = 0.94, SE = 0.004; SAC = 0.93, SE = 0.003).

Migrant Birds

Year-watershed interactions existed for numbers of individuals per plot (F = 4.32; df = 9, 2035; P < 0.001), species per plot (F = 3.54; df = 9, 2035; P < 0.001), and diversity per plot (F = 3.02; df = 9, 2035; P = 0.001). Thus, these variables were interpreted within years. Numbers of individuals per plot, species per plot, and diversity per plot were consistently highest for migrants in LG, followed by NAC, each of the 4 years (table 4). Similar numbers were recorded on BC and SAC each year. SAC had the lowest species per plot and diversity per plot in 1995 and 1998. No year-watershed interaction was found for evenness per plot (F = 1.29; df = 9, 1759; P = 0.236). Evenness per plot was highest in

Table 4—Mean (SE) breeding bird community metrics for migrant species recorded on four watersheds under different levels of forest management activity in the Ouachita Mountains of Arkansas

	Year			
Watershed	1995	1996	1997	1998
	number of individuals per plot			
Little Glazypeau	6.27 A[a] (0.336)	9.91 A (0.466)	8.61 A (0.332)	7.87 A (0.354)
North Alum Creek	4.53 B (0.203)	6.44 B (0.252)	7.81 A (0.279)	7.00 B (0.289)
Bread Creek	3.27 C (0.256)	4.16 C (0.287)	5.65 B (0.471)	5.14 C (0.426)
South Alum Creek	2.40 C (0.156)	3.61 C (0.203)	5.30 B (0.277)	3.69 D (0.314)
	F = 33.19 df = 3, 498 P < 0.001	F = 59.76 df = 3, 492 P < 0.001	F = 18.50 df = 3, 539 P < 0.001	F = 22.30 df = 3, 506 P < 0.001
	number of species per plot			
Little Glazypeau	3.80 A (0.080)	5.44 A (0.199)	5.52 A (0.180)	4.53 A (0.163)
North Alum Creek	2.94 B (0.117)	3.64 B (0.134)	4.61 B (0.151)	4.01 B (0.153)
Bread Creek	2.31 C (0.173)	2.62 C (0.176)	3.21 C (0.237)	3.24 C (0.228)
South Alum Creek	1.69 D (0.092)	2.15 C (0.102)	3.18 C (0.140)	2.22 D (0.134)
	F = 30.29 df = 3, 498 P < 0.001	F = 64.04 df = 3, 492 P < 0.001	F = 32.43 df = 3, 539 P < 0.001	F = 25.00 df = 3, 506 P < 0.001
	diversity per plot			
Little Glazypeau	1.13 A (0.048)	1.49 A (0.043)	1.53 A (0.038)	1.32 A (0.039)
North Alum Creek	0.87 B (0.037)	1.06 B (0.037)	1.30 B (0.035)	1.13 B (0.041)
Bread Creek	0.64 C (0.068)	0.76 C (0.062)	0.91 C (0.071)	0.94 C (0.069)
South Alum Creek	0.43 D (0.045)	0.63 C (0.045)	1.00 C (0.041)	0.60 D (0.056)
	F = 31.88 df = 3, 498 P < 0.001	F = 56.26 df = 3, 492 P < 0.001	F = 34.67 df = 3, 539 P < 0.001	F = 29.77 df = 3, 506 P < 0.001

[a] Means in a column followed by the same letter did not differ (P > 0.05).

LG ($\bar{x}$ = 0.94, SE = 0.002) and BC ($\bar{x}$ = 0.94, SE = 0.004) and lowest in NAC ($\bar{x}$ = 0.93, SE = 0.002) and SAC ($\bar{x}$ = 0.92, SE = 0.005) (F c= 5.72; df = 3, 9; P = 0.018).

Residents

No year-watershed interactions existed for numbers of individuals per plot (F = 1.17; df = 9, 1760; P = 0.310), species per plot (F = 1.83; df = 9, 1760; P = 0.059), diversity per plot (F = 1.74; df = 9, 1760; P = 0.074), or eveness per plot (F = 1.18; df = 9, 1056; P = 0.303). LG had the highest individuals per plot (F = 25.53; df = 3, 9; P < 0.001), species per plot (F = 32.05; df = 3, 9; P < 0.001), and diversity per plot (F = 28.75; df = 3, 9; P < 0.001) of all watersheds (table 5). NAC, BC, and SAC were comparable for most community metrics. Evenness did not differ among watersheds (F = 2.19; df = 3, 9; P = 0.152).

Cavity Nesters

No year-watershed interactions existed for numbers of individuals per plot (F = 1.63; df = 9, 1201; P = 0.102), species per plot (F = 1.81; df = 9, 1201; P = 0.062), or diversity per plot (F = 1.69; df = 9, 1201; P = 0.087). There were no differences among watersheds for numbers of individuals per plot (F = 2.25; df = 3, 9; P = 0.150), species per plot (F = 2.81; df = 3, 9; P = 0.099), or diversity per plot (F = 2.95; df = 3, 9; P = 0.089) (table 6). A year-watershed interaction was found for evenness per plot (F = 2.72; df = 9, 457; P = 0.004), and evenness per plot did not differ between watersheds during 1995 (F = 2.40; df = 3, 90; P = 0.073), 1997 (F = 1.48; df = 3, 180; P = 0.223), or 1998 (F = 0.81; df = 3, 103; P = 0.494). Evenness per plot did differ in 1996 (F = 4.79; df = 3, 84; P = 0.004), with SAC having a lower value than all other watersheds.

Canopy Nesters

Year-watershed interactions existed for numbers of individuals per plot (F = 2.55; df = 9, 1963; P = 0.007), species per plot (F = 1.99; df = 9, 1963; P = 0.037), and diversity per plot (F = 1.87; df = 9, 1963; P = 0.050). Thus, these variables were interpreted within years. In 1995 and 1997, there were no differences in numbers of individuals per plot between watersheds (table 7). Numbers of individuals per plot were highest in LG in 1996 and lowest in BC in 1998. In 1995 and 1996, species per plot was highest in LG, with no differences in the other watersheds. Species per plot was lowest in SAC in 1998. Diversity per plot was highest in LG in 1996, with inconsistent differences in other years. No year-watershed interaction was found for evenness per plot

Table 5—Mean (SE) breeding resident bird community metrics recorded on four watersheds under different levels of forest management activity in the Ouachita Mountains of Arkansas

Watershed	Individuals per plot	Species per plot	Diversity per plot	Evenness per plot
	- - - - - - - - - - - - - *number*	- - - - - - - - - - - - -		
Little Glazypeau	4.21 A[a]	2.77 A	0.81 A	0.94
	(0.238)	(0.253)	(0.263)	(0.267)
North Alum Creek	2.95 B	1.96 B	0.51 B	0.94
	(0.172)	(0.100)	(0.159)	(0.190)
Bread Creek	2.76 B	1.79 C	0.43 C	0.95
	(0.211)	(0.165)	(0.242)	(0.279)
South Alum Creek	3.08 B	1.83 BC	0.44 C	0.93
	(0.263)	(0.194)	(0.255)	(0.237)

[a] Means in a column followed by the same letter did not differ (P > 0.05).

Table 6—Mean (SE) cavity nesting bird community metrics recorded on four watersheds under different levels of forest management activity in the Ouachita Mountains of Arkansas

Watershed	Individuals per plot	Species per plot	Diversity per plot	Evenness per plot
	- - - - - - - - - - - - *number*	- - - - - - - - - - - -		
Little Glazypeau	2.29	1.57	0.34	0.94
	(0.080)	(0.041)	(0.022)	(0.010)
North Alum Creek	2.02	1.48	0.29	0.92
	(0.064)	(0.032)	(0.017)	(0.013)
Bread Creek	1.89	1.33	0.20	0.93
	(0.115)	(0.051)	(0.029)	(0.027)
South Alum Creek	2.07	1.38	0.25	0.85
	(0.110)	(0.051)	(0.027)	(0.034)

Table 7—Mean (SE) canopy nesting bird community metrics recorded on four watersheds under different levels of forest management activity in the Ouachita Mountains of Arkansas

	Year			
Watershed	1995	1996	1997	1998
		number of individuals per plot		
Little Glazypeau	3.13	5.29 A[a]	5.42	4.84 A
	(0.190)	(0.654)	(0.268)	(0.248)
North Alum Creek	2.82	4.05 B	4.92	4.89 A
	(0.134)	(0.289)	(0.186	(0.194)
Bread Creek	2.58	3.59 B	4.86	4.30 AB
	(0.189)	(0.357)	(0.430)	(0.310)
South Alum Creek	2.60	3.65 B	5.57	3.94 B
	(0.203)	(0.316)	(0.262)	(0.225)
	$F = 1.68$	$F = 8.53$	$F = 1.62$	$F = 3.13$
	df = 3, 449	df = 3, 477	df = 3, 531	df = 3, 506
	$P = 0.170$	$P < 0.001$	$P = 0.184$	$P = 0.025$
		number of species per plot		
Little Glazypeau	2.08 A	2.89 A	3.32 A	2.72 A
	(0.103)	(0.290)	(0.129)	(0.112)
North Alum Creek	1.80 B	2.27 B	2.78 BC	2.53 A
	(0.062)	(0.170)	(0.088)	(0.075)
Bread Creek	1.77 B	2.13 B	2.53 B	2.45 AB
	(0.096)	(0.230)	(0.140)	(0.127)
South Alum Creek	1.65 B	2.01 B	2.97 AC	2.22 B
	(0.093)	(0.170)	(0.109)	(0.114)
	$F = 3.84$	$F = 12.09$	$F = 7.38$	$F = 3.39$
	df = 3, 449	df = 3, 477	df = 3, 531	df = 3, 506
	$P = 0.010$	$P < 0.001$	$P < 0.001$	$P = 0.018$
		diversity per plot		
Little Glazypeau	0.58 A	0.83 A	1.00 A	0.81 A
	(0.047)	(0.041)	(0.042)	(0.042)
North Alum Creek	0.47 B	0.66 B	0.83 BC	0.74 A
	(0.030)	(0.035)	(0.031)	(0.029)
Bread Creek	0.46 AB	0.59 B	0.74 C	0.72 AB
	(0.052)	(0.061)	(0.055)	(0.050)
South Alum Creek	0.41 B	0.54 B	0.94 AB	0.63 B
	(0.045)	(0.052)	(0.038)	(0.049)
	$F = 2.65$	$F = 7.39$	$F = 6.47$	$F = 2.98$
	df = 3, 449	df = 3, 477	df = 3, 531	df = 3, 506
	$P = 0.049$	$P < 0.001$	$P < 0.001$	$P = 0.031$

[a] Means in a column followed by the same letter did not differ ($P > 0.05$).

($F = 0.85$; df = 9, 1480; $P = 0.570$), and evenness per plot did not differ ($F = 0.973$; df = 3, 9; $P = 0.445$) among watersheds (LG = 0.92, SE = 0.004; NAC = 0.92, SE = 0.003; BC = 0.92, SE = 0.006; SAC = 0.92, SE = 0.005).

Ground and Shrub Nesters

A year-watershed interaction existed for numbers of individuals per plot ($F = 1.99$; df = 9, 1628; $P = 0.037$); thus, this variable was interpreted within years. Numbers of individuals per plot were consistently highest in LG and lowest in SAC over all years (table 8). No year-watershed interactions existed for species per plot ($F = 0.93$; df = 9, 1628; $P = 0.499$), diversity per plot ($F = 0.84$; df = 9, 1628; $P = 0.582$), or evenness per plot ($F = 0.55$; df = 9, 1035; $P = 0.841$). Species per point and diversity per point were consistently highest in LG and lowest in SAC over all years (table 9). Evenness did not differ among watersheds.

DISCUSSION AND CONCLUSIONS

Because this study focused on differences and similarities of breeding bird community characteristics among watersheds, replication was not feasible. Thus, our inferences and conclusions are limited to the bird communities and landscapes contained within the specific watersheds we studied.

Table 8—Mean (SE) numbers of individual ground and shrub nesting birds per plot recorded on four watersheds under different levels of forest management activity in the Ouachita Mountains of Arkansas

	Year			
Watershed	1995	1996	1997	1998
Little Glazypeau	5.13 A[a]	7.26 A	6.04 A	5.99 A
	(0.409)	(0.654)	(0.484)	(0.485)
North Alum Creek	3.41 B	4.08 B	5.05 B	4.76 B
	(0.234)	(0.289)	(0.354)	(0.422)
Bread Creek	2.66 B	2.98 BC	4.32 B	3.50 C
	(0.350)	(0.357)	(0.652)	(0.533)
South Alum Creek	1.19 C	1.85 C	2.59 C	2.47 C
	(0.083)	(0.316)	(0.413)	(0.429)
	$F = 21.53$	$F = 29.99$	$F = 14.34$	$F = 11.96$
	df = 3, 364	df = 3, 392	df = 3, 467	df = 3, 405
	$P = < 0.001$	$P < 0.001$	$P < 0.001$	$P = < 0.001$

[a] Means in a column followed by the same letter did not differ ($P > 0.05$).

Table 9—Mean (SE) ground and shrub nesting bird community metrics recorded on four watersheds under different levels of forest management activity in the Ouachita Mountains of Arkansas

Watershed	Species per plot	Diversity per plot	Evenness per plot
	no.		
Little Glazypeau	3.70 A	1.07 A	0.94
	(0.094)	(0.027)	(0.003)
North Alum Creek	2.75 B	0.73 B	0.94
	(0.075)	(0.024)	(0.003)
Bread Creek	2.27 C	0.57 C	0.96
	(0.123)	(0.044)	(0.004)
South Alum Creek	1.41 D	0.25 D	0.94
	(0.053)	(0.024)	(0.010)

[a] Means in a column followed by the same letter did not differ ($P > 0.05$).

The differing ownerships, management objectives, and intensities of forest management activity among watersheds influenced breeding bird communities. Spatial and compositional habitat heterogeneity was highest in LG (Tappe and others, in press) and was reflected in higher numbers of individuals per point, species per point, and diversity per point for both migrants and residents. Lower values were seen in BC and SAC, with NAC being intermediate. Thus, these patterns appear to follow Tappe and others (in press) ranking of spatial heterogeneity in the watersheds (LG > NAC > BC > SAC).

Patterns of community characteristics within nesting guilds were not as consistent as when birds were separated into migrant and resident species. Though somewhat variable, watersheds were comparable during most years in respect to canopy nesters. Community characteristics of cavity nesters were the same for all watersheds. Each watershed provided some suitable habitat for species in these nesting guilds. Ground and shrub nesters had their lowest numbers of individuals per point, species per point, and diversity per point in SAC and their highest in LG, with NAC and BC being intermediate to these values. Values for ground and shrub nesters reflected the amount of suitable, early successional habitat found in each watershed. LG was composed of approximately 22 percent young/open and thin/open pine stands whereas SAC was composed of approximately 1 percent young/open and thin/open pine stands (Tappe and others, in press).

Each watershed was characterized by differing breeding bird community structure. LG provided a variety of early-successional and medium-aged habitats. Several species prefer these habitats, many of which are species of concern, including the black and white warbler, prairie warbler (*Dendroica discolor*), and worm-eating warbler (*Helmitheros vermivorus*). SAC provided a relatively homogenous landscape of more mature forest, thus benefitting species such as the ovenbird and scarlet tanager (*Piranga olivacea*). NAC and BC both provided landscapes that included differing proportions of each of these habitats, supporting differing bird communities. Thus, the spatial structure and composition of each watershed resulted in differing breeding bird community associations.

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AMPHIBIAN COMMUNITIES UNDER DIVERSE FOREST MANAGEMENT IN THE OUACHITA MOUNTAINS, ARKANSAS

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Abstract—From May 1995 to March 1999, we censused amphibians in the Ouachita Mountains, Arkansas, on 60 plots on each of four forested watersheds five times per year, with new plots each year. We found negligible differences in species richness among watersheds, and community similarities were high, even though most pairwise comparisons were significantly different. The two most intensively managed watersheds were more similar to each other than to those less intensively managed, and the former had nominally higher overall species evenness and (beta) diversity. At the plot level, we found no significant differences in abundance, species richness, or (alpha) diversity, although the two most intensively managed watersheds had plots with fewer species and less diversity. Detrended correspondence analysis revealed that communities diverged on the basis of presumed gradients of terrestrial-aquatic, elevation, and canopy cover. Important communities that differed among watersheds were those of small, often ephemeral ponds and large, permanent ponds.

INTRODUCTION

Little is known about habitat parameters that influence amphibian communities, and even less is known about landscape-level environmental influences (Dodd and Cade 1998). Moreover, many species of amphibians are declining worldwide in abundance, and more baseline data on natural populations over time are needed to appraise the suspected causes.

Amphibian populations, particularly anurans, are catastrophically declining worldwide (Blaustein and Wake 1995, Houlahan and others 2000, Phillips 1990, Wake 1991). Habitat loss, or modification to the extent that it is only marginally suitable for amphibians, is considered by most biologists to be the principal cause for the decline of amphibian populations and species (Blaustein and Wake 1995). It has been suggested that local environmental degradation is insidiously reducing amphibian diversity (Delis and others 1996). Examples of human activities that might result in such alterations include impoundment of natural waterways, mining, agriculture, urbanization, and forest management.

We report results of pretreatment data collection for a large-scale, long-term, field study of amphibian communities and the influence of habitat and landscape environmental variables in four watersheds of the Ouachita Mountains of west-central Arkansas. Following the pretreatment stage, sections of the watersheds will be subjected to different forest management to achieve a variety of specific “desired future conditions.” After treatment, data on amphibian communities will again be collected and used to quantify community changes and to compare with the predictions of multivariate community models that we are developing. This long-term study is one component of Phase III of the Ouachita Mountains Ecosystem Management Research Project; the wildlife component of this cooperative effort involves Weyerhaeuser Company, the National Council of the Paper Industry for Air and Stream Improvement, Oklahoma State University, the Oklahoma Cooperative Fish and Wildlife Research Unit, the University of Arkansas Monticello, the Ouachita National Forest, and the Southern Research Station of the USDA Forest Service.

The objectives of our overall study are to: (1) characterize reptile and amphibian communities in four watersheds representing markedly different forest-management strategies in the Ouachita Mountains, Arkansas; (2) develop and validate models for predicting community composition based on site, stand, and landscape parameters; and (3) develop recommendations to promote maintenance of reptile and amphibian communities in managed forest landscapes. This report contains results for amphibians of the pretreatment data analysis performed at the end of four survey years.

METHODS

Study Areas

The study was conducted on four 1500- to 4000-ha watersheds under different intensities of management in Garland and Saline counties near Hot Springs, AR. The watersheds differed markedly with respect to factors such as mean rotation lengths, forest type diversity, stand sizes and ages, and the amount of natural second-growth coverage (Guldin and others, Tappe and others, in press).

Little Glazypeau, a watershed located some 22 km southwest of the other three watersheds (that were contiguous) and managed largely for sawlog production by Weyerhaeuser Company, represented our most intensively managed

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watershed. Much of the second-growth shortleaf pine (*Pinus echinata*)-hardwood forest that originally covered this watershed had been harvested and planted to loblolly pine (*P. taeda*) plantations of 9-142 ha. Typically, these plantations were thinned twice, pruned to 5-8 m high, fertilized, and harvested at 30-35 years old. The remaining, selectively-harvested acreage in the watershed occured on rocky ridgetops, steep slopes, and streamside management zones that were retained for watershed protection and to provide habitat diversity for wildlife. South Alum, an experimental section of the Ouachita National Forest that has received minimal logging for > 80 years, represented the least intense level of silviculture. South Alum was almost entirely USDA Forest Service and consisted of mature forest over most of the area. Bread Creek and North Alum fell in between these extremes of forest management. From independent records of forest management and present composition of number, age, and distribution of pine (mostly *Pinus echinata*) plantations, Bread Creek was considered less intensely managed than North Alum (Tappe and others, in press). Bread Creek was primarily USDA Forest Service land, and had been managed according to prevailing Forest Service standards and guidelines for several decades, whereas North Alum was of mixed ownership, with about half of the area under Weyerhaeuser management and half under USDA Forest Service management. North Alum displayed characteristics of a diverse range of management activities, ranging from no management along steeper slopes and higher elevations to intensively managed pine plantations, mainly at lower elevations. Thus, the watersheds, in order of intensity of forest management, ran South Alum, Bread Creek, North Alum, and Little Glazypeau. These same watersheds, in order of size, were South Alum (1500 ha), Bread Creek (1535 ha), Little Glazypeau (2273 ha), and North Alum (3961 ha).

Sampling Plots

We surveyed amphibians and reptiles using area-constrained searches on a subset (56) of the 75-235 plots/watershed that were established each year for breeding bird surveys (Tappe and others, in press). Bird plots were established at 200-m intervals along > 100 km of parallel transects (oriented approximately north-south across prevailing topography) that were established in 1995 over the 4 watersheds. These same transects were used in 1996, but new plots were established by shifting plot centers 100 m. In 1997, approximately 110 km of new transects were established between those of 1995; new plots were established in 1998 along these new transects as in 1996. The subset of plots that were used for amphibian surveys were selected to represent a cross-section of slopes, aspects, forest types, stand conditions, and aquatic habitats. The center of our 20-m-radius (0.13-ha) plots also was the center of a bird sampling plot. Plastic flagging was used to delineate plot boundaries on all amphibian plots. In each watershed, we selected amphibian plots to make sure there were at least 12-15 of them in aquatic habitat, which consisted of springs, streams, and man-made ponds that had been established to benefit wildlife (Forest Service wildlife ponds) and/or as sources of water for fire fighting. An additional four plots per watershed per year were established off the transects at these ponds (or at wide pools in the high-order streams at the bottom of a watershed) to ensure that we had equivalent sampling effort at these aquatic habitats. Plots at these aquatic sites were established so that the center of the plot was on land right next to the shore (roughly half of the plot was over water and half over land). Thus, we surveyed 60 plots per year per watershed, in total.

Amphibian Surveys

Trained crews of 3-5 individuals surveyed each set of 60 plots per watershed during daylight hours 5 times a year from May 1995 to March 1999: early May, late May, mid-June, early October, and early the following March. Plots were surveyed entirely by visually searching vegetation and the ground surface, and by lifting cover objects (rocks, logs, and debris); the latter were replaced to minimize impacts on subsequent surveys. We sampled plots in deep water by having one person move slowly through the water in chest waders and carefully scan the water and shoreline for animals. Some animals were captured in this way by use of a dipnet. Amphibians that were seen and identified were tallied; those that required capture for identification were released at the point of capture before leaving the plot. Additionally, any anurans calling on the plot, but not seen, were identified by their call and counted. Anuran species calling off-plot in similar habitat were recorded as present, but no attempt was made to count the number of individuals.

Data Analysis

Amphibian count data were pooled across the five sampling periods per year, but data from each plot served as separate samples. Data collected in the first year (1995) from South Alum and North Alum were excluded from analyses due to differences in sampling effort by former collaborators, and data from a few other plots were discarded when five surveys per year were not attained. Thus, analyses presented here are based on data from 833 plots, each censused five times, for a total of 4,165 censuses.

Watershed level—Community indices were first calculated at the watershed level: number of individuals by species for all years combined, amphibian abundance, species richness, (beta) species diversity [H'; Shannon-Wiener diversity index (Shannon and Weaver 1949)], and species evenness [diversity divided by maximal diversity, or H'/(ln number of species); Pielou 1966]. With only four watersheds to compare, no statistical tests were employed. Because the Shannon-Wiener diversity index is quite sensitive to sample size and because we did not have an equal effort among all watersheds (two watersheds with data from only three years; see previous statement), we randomly reordered the plots within each watershed, pooling years, and plotted the Shannon-Wiener diversity index of each watershed for cumulative sets of plots up to the total number of plots for each watershed. Such a plot would show if diversity approached an asymptote as cumulative plots increased and if our total number of plots per watershed was sufficient to adequately estimate species diversity.

We computed Morisita's index of community similarity (Morisita 1959) between all pairs of watersheds, pooling data for the entire study for each watershed. This index is desirable because sample sizes and species diversities of the communities being compared have little influence on its calculation (Morisita 1959; Wolda 1981). Using additive tree cluster analysis (Sattath and Tversky 1977), we converted that matrix of similarities into a dendrogram of communities such that the total length of branches connecting any two communities was proportional to the compositional difference between those communities. To statistically compare the various Morisita's indices of community similarity, we conducted randomization tests (Biondini and others 1988). For each pair of watersheds, we randomly reassigned plots between them (retaining the sample size of each watershed) and computed Morisita's index for these two "synthetic" communities. We repeated this procedure 1,000 times and tabulated the number of times the recomputed index was smaller than or equal to the "actual" index. If less than 100 recomputed indices fell below or equal to the "actual" index, then those two watersheds were considered different ($p < 0.10$) in species composition.

Plot level—For analyses at the plot level, we used mixed model, two-way ANOVAs (ANOVAs with both random and fixed effects; PROC MIXED, SAS 1999) to test for differences among watersheds, years, and year * watershed interaction. We recognize that our sample of plots drawn from each of four watersheds is pseudoreplication (Hurlbert, 1984), but the large scale of this study preempted the sampling of a sufficient number of replicate watersheds per treatment class. While the results of our ANOVA must be interpreted with caution due to this pseudoreplication, we feel that the analysis nevertheless suggests likely ecological patterns that deserve attention. The response variables of the ANOVAs were (1) amphibian abundance per plot, (2) species richness per plot, and (alpha) plot diversity (Shannon-Wiener index). We first transformed the count variables (1 and 2) by the square root, SQR (count + 0.5), to make distributions within cells of the ANOVA more normally distributed. Even with those transformations, our data did not fully meet assumptions of normality and homogeneity of variances, but the Satterthwaite algorithm of the mixed model ANOVA is relatively robust to abuses of these assumptions, especially of homogeneity of variances (SAS 1999), and so we proceeded with these parametric analyses. We recognized differences in weather between years, not of interest to us here, and included year effects and year * watershed interaction as random effects, not to be statistically interpreted. The fixed factor (watersheds) was tested for statistical significance at $p < 0.10$. If a significant watershed effect was found, we used LSD to evaluate pairwise differences between any two watersheds.

Detrended canonical correspondence analysis (DCA)—To appraise amphibian community composition, develop preliminary hypotheses of presumptive environmental gradients influencing these communities, and compare graphically the environments and amphibian communities of the four watersheds, we used DCA (ter Braak and Prentice 1988, ter Braak and Šmilauer 1998), pooling plots from all four watersheds and years. Application of DCA to our data allowed for a more detailed inspection of amphibian communities at the plot level and how they were distributed along inferred environmental gradients. DCA is probably the most widely employed eigenanalysis-based ordination technique used by community ecologists. It is an indirect ordination method that orders plots with similar compositions of species along multiple axes simultaneously. The statistical algorithm is to calculate sample scores of each plot as a weighted average of the species scores, and species scores as a weighted average of samples scores; iterations are repeated until there is no further change in scores, at which time samples (plots) with similar animal communities appear clustered when plotted on multiple axes. Environmental gradients are inferred from the pattern of species and/or plots and the biologist's knowledge of the species. As a step beyond single-number summary statistics like diversity indices or Morisita's indices, DCA results in a cloud of points for separate species in n-dimensional space, conventionally viewed as centroids (averaged central tendency) in two dimensions at a time. In other words, DCA results in a pattern, not a number. For our analysis, rare species (less than three individuals encountered over all four watersheds for all four years) and plots where no amphibian species were found were excluded due to computational constraints.

RESULTS AND DISCUSSION

We found 4,214 individuals of 20 amphibian species during our four-year study (table 1). Total species diversity was 1.76 for all watersheds pooled. We found a mean of 5.06 amphibians per plot (median = 3.00, range = 0-64).

Watershed Level

Despite large differences in size of watersheds and substantial differences in management intensities, the amphibian communities of these four watersheds were fairly similar. Species richness was just 17-18 species per watershed, and 15 of the total set of 20 species (75 percent) were common to all four watersheds (table 1). The watersheds differed some in both overall species diversity and evenness; the order from lowest to highest by both measures was Bread Creek, South Alum, Little Glazypeau, and North Alum (table 1). Bread Creek and South Alum were virtually identical in these measures. Recalculated diversity indices against cumulative plots showed that diversity leveled off after about 40-100 plots (fig. 1), well below the lowest total of plots for any watershed. North Alum showed distinctly higher overall diversity than the other watersheds, and in rank order, the two watersheds most intensively managed had higher species diversity than the two least intensively managed, although Little Glazypeau was not that different from Bread Creek or South Alum.

By far, the dominant species found in all watersheds was the western slimy salamander (*Plethodon albagula*), representing on the average 60.0 percent of each watershed community. Additionally, species composition of the six most ubiquitous species of each watershed was strikingly similar (table 2).

We examined the overall set of species to see if there were any species absent from all but one watershed, or present in only one watershed. One species, the central newt (*Notophthalmus viridescens*), was absent from Little Glazypeau; and one species, the Strecker's chorus frog (*Pseudacris streckeri*), was present only in Little Glazypeau, although represented by only one observed individual (table 1).

Table 1—Amphibian abundance on four watersheds in the Ouachita Mountains, Arkansas, 1995-1999

Species	Little Glazypeau	North Alum	Bread Creek	South Alum
	- - - - - - - - - - - - -	*number*	- - - - - - - - - - - -	
Acris crepitans				
Cricket frog	238	143	48	3
Ambystoma annulatum				
Ringed salamander	0	0	3	1
Ambystoma maculatum				
Spotted salamander	0	1	4	0
Bufo americanus				
American toad	44	56	189	49
Bufo woodhousei				
Woodhouse's toad	2	2	7	2
Desmognathus brimleyorum				
Ouachita dusky salamander	74	17	15	9
Eurycea multiplicata				
Many-ribbed salamander	17	33	68	48
Gastrophryne carolinensis				
Eastern narrowmouth toad	24	22	10	1
Hemidactylium scutatum				
Four-toed salamander	14	6	2	2
Hyla chrysoscelis				
Gray treefrog	39	17	43	15
Hyla cinerea				
Green treefrog	1	1	0	0
Notophthalmus viridescens				
Central newt	0	7	3	2
Plethodon albagula				
Western slimy salamander	642	355	890	368
Plethodon serratus				
Southern redback salamander	13	22	57	195
Pseudacris crucifer				
Spring peeper	11	39	11	2
Pseudacris streckeri				
Strecker's chorus frog	1	0	0	0
Pseudacris triseriata				
Upland chorus frog	33	26	20	10
Rana catesbeiana				
Bullfrog	6	6	2	3
Rana clamitans				
Green frog	53	30	70	38
Rana utricularia				
Southern leopard frog	7	9	12	1
Total	1,219	792	1,454	749
Species richness	17	18	18	17
Species diversity	1.65	1.94	1.50	1.53
Species evenness	0.58	0.67	0.51	0.53

Data are arrayed (left to right) from the most to the least intensively managed watersheds.

Using Morisita's index of community similarity, watershed amphibian communities were quite similar, ranging from 0.84 to 0.98 (table 3). South Alum differed the most from the other watersheds, displaying an index of 0.84 with both Little Glazypeau and North Alum. The additive tree cluster analysis grouped the two most intensively managed watersheds (Little Glazypeau and North Alum) and the two least intensively managed watersheds (Bread Creek and South Alum) (fig. 2). Despite generally high indices of community similarity, all of the indices [except that between North Alum and Little Glazypeau (0.98)] were statistically significant by the randomization tests; i.e., all watershed pairs except this one were significantly different from each other beyond that expected by chance.

Table 2—Six most common amphibian species found in each watershed in order of decreasing abundance

Little Glazypeau	n	North Alum	n	Bread Creek	n	South Alum	n
Plethodon albagula (Slimy salamander)	642	*Plethodon albagula* (Slimy salamander)	355	*Plethodon albagula* (Slimy salamander)	890	*Plethodon albagula* (Slimy salamander)	368
Acris crepitans (Cricket frog)	238	*Acris crepitans* (Cricket frog)	143	*Bufo americanus* (American toad)	189	*Plethodon serratus* (Redback salamander)	195
Desmognathus brimleyorum (Ouachita dusky salamander)	74	*Bufo americanus* (American toad)	56	*Rana clamitans* (Green frog)	70	*Bufo americanus* (American toad)	49
Rana clamitans (Green frog)	53	*Pseudacris crucifer* (Spring peeper)	39	*Eurycea multiplicata* (Many ribbed salamander)	68	*Eurycea multiplicata* (Many ribbed salamander)	48
Bufo americanus (American toad)	44	*Eurycea multiplicata* (Many ribbed salamander)	33	*Plethodon serratus* (Redback salamander)	57	*Rana clamitans* (Green frog)	38
Hyla chrysoscelis (Gray treefrog)	39	*Rana clamitans* (Green frog)	30	*Acris crepitans* (Cricket frog)	48	*Hyla chrysoscelis* (Gray treefrog)	15

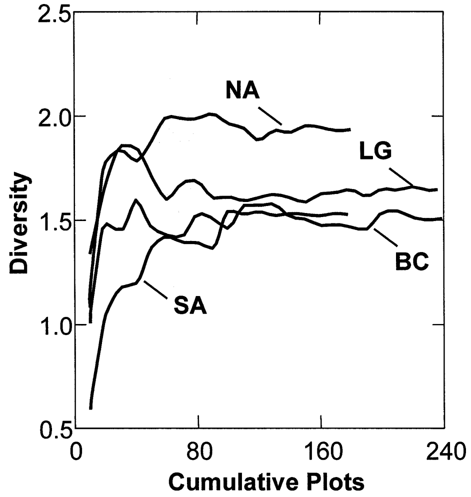

Figure 1—Diversity of each watershed against cumulative number of plots (in random order) included in recalculation (SA = South Alum, BC = Bread Creek, NA = North Alum, LG = Little Glazypeau).

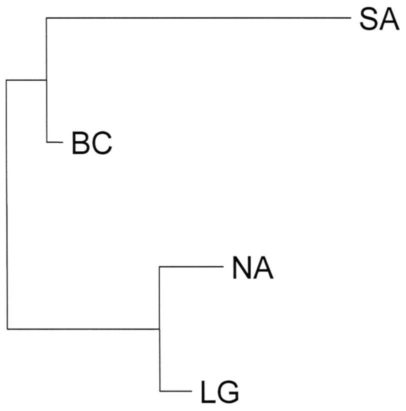

Figure 2—Additive tree dendrogram of amphibian community similarity of four watersheds constructed from pairwise Morisita Indices (SA = South Alum, BC = Bread Creek, NA = North Alum, LG = Little Glazypeau). Difference between any two communities is proportional to the total branch length connecting them.

Table 3—Morisita's index of community similarity for all pairs of watersheds

Watershed	Little Glazypeau	North Alum	Bread Creek	South Alum
Little Glazypeau	1.00			
North Alum	0.98	1.00		
Bread Creek	0.94[a]	0.92[a]	1.00	
South Alum	0.84[a]	0.84[a]	0.91[a]	1.00

Index ranges from 0 to 1, where 1 means communities are identical.

[a] Significantly dissimilar by Randomization test, $p < 0.10$.

Plot Level

We found no differences in total number of individuals per plot, species richness per plot, or species diversity per plot between the watersheds (table 4), as indicated by ANOVA. Consequently, we did not perform pairwise contrasts.

Detrended Correspondence Analysis

Detrended correspondence analysis indicated relatively long environmental gradients along the first three axes for amphibian communities in the four watersheds (table 5). The fourth and additional axes contributed little to the pattern of community organization since each additional

Table 4—Number of plots surveyed (1995–1999), mean amphibian abundance per plot, mean species richness per plot, and mean species diversity per plot by watershed

Watershed	*n*	Mean abundance[a]	Mean species richness[b]	Mean species diversity[c]
Little Glazypeau	236	5.17	1.35	0.25
North Alum	180	4.40	1.51	0.32
Bread Creek	239	6.08	1.90	0.43
South Alum	178	4.21	1.70	0.41

[a] ANOVA: $F_3 = 1.73$, $p = 0.25$.
[b] ANOVA: $F_3 = 2.61$, $p = 0.14$.
[c] ANOVA: $F_3 = 2.56$, $p = 0.14$.

Table 5—Eigenvalues of first 4 axes of detrended correspondence analysis of 18 amphibian species distributed among a pooled total of 681 plots on the 4 watersheds[a]

Axis	Eigenvalue
One	0.795
Two	0.513
Three	0.430
Four	0.282

[a] An eigenvalue is the correlation coefficient between the plot scores and species scores along a given axis where each axis is orthogonal (independent) to all previous axes in the analysis.

axis explains only residual variation not already incorporated into the DCA. In other words, plots and species scores were relatively tightly correlated with each other along an appreciable stretch of at least the first three axes (eigenvalues range from 0 to 1: high eigenvalues mean that clouds of points are spread linearly along each axis, and low eigenvalues mean that points are clustered at the center of each axis). Species' centroids plotted against axes two vs. one (fig. 3) and against axes three vs. one (fig. 4), showed strong separation of species.

The pattern of species' centroids along axis one suggested that it was a moisture gradient from terrestrial (left) to aquatic conditions (right) (fig. 3). Those species scoring the lowest on this axis were the very terrestrial woodland salamanders, *Plethodon serratus* (southern redback salamander) and *P. albagula* (western slimy salamander). Further to the right were the more aquatic salamanders, *Desmognathus brimleyorum* (Ouachita dusky salamander) and *Eurycea multiplicata* (many-ribbed salamander), which frequently were found in small streams or shallow backwaters of larger water courses. Next to the right along axis one were the community of amphibians of small, generally fishless, often ephemeral, ponds (*Bufo americanus*, *Hyla chrysoscelis*, *Ambystoma maculatum*, *Notophthalmus viridescens*, and

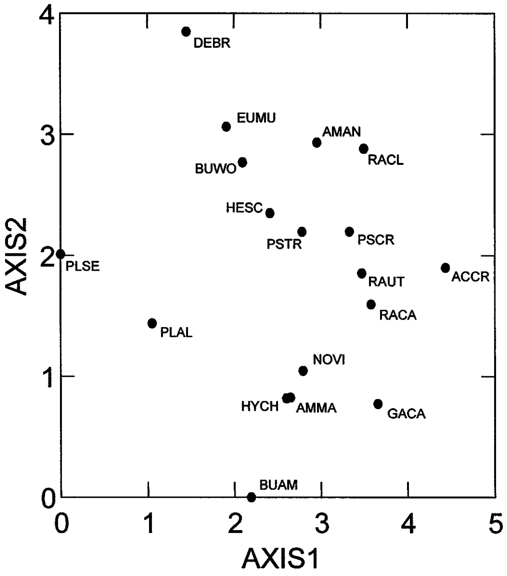

Figure 3—Centroids of species' scores from detrended correspondence analysis (DCA) of censused plots on all four watersheds pooled: DCA axis two vs. one (ACCR = *Acris crepitans*; AMAN = *Ambystoma annulata*; AMMA = *A. maculatum*; BUAM = *Bufo americanus*; BUWO = *B. woodhousei*; DEBR = *Desmognathus brimleyorum*; EUMU = *Eurycea multiplicata;* GACA = *Gastrophryne carolinensis*; HESC = *Hemidactylium scutatum*; HYCH = *Hyla chrysoscelis*; NOVI = *Notophthalmus viridescens*; PLAL = *Plethodon albagula*; PLSE = *P. serratus*; PSCR = *Pseudoacris crucifer*, PSTR = *P. triseriata*; RACA = *Rana catesbeiana*; RACL = *R. clamitans*; RAUT = *R. utricularia*).

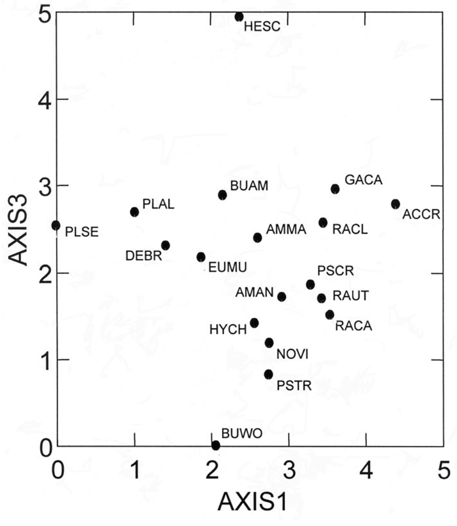

Figure 4—Centroids of species' scores from detrended correspondence analysis (DCA) of censused plots on all four watersheds pooled: DCA axis three vs. one (ACCR = *Acris crepitans*; AMAN = *Ambystoma annulata*; AMMA = *A. maculatum*; BUAM = *Bufo americanus*; BUWO = *B. woodhousei*; DEBR = *Desmognathus brimleyorum*; EUMU = *Eurycea multiplicata;* GACA = *Gastrophryne carolinensis*; HESC = *Hemidactylium scutatum*; HYCH = *Hyla chrysoscelis*; NOVI = *Notophthalmus viridescens*; PLAL = *Plethodon albagula*; PLSE = *P. serratus*; PSCR = *Pseudoacris crucifer*, PSTR = *P. triseriata*; RACA = *Rana catesbeiana*; RACL = *R. clamitans*; RAUT = *R. utricularia*).

Gastrophryne carolinensis; American toad, gray treefrog, spotted salamander, central newt, and eastern narrowmouth toad). Aligned vertically with this community but extending toward more extreme values of axis one were mapped the community of salamanders and frogs that were most aquatic and associated with small to large ponds and quiet pools of larger streams (*Ambystoma annulata, B. woodhousei*, *Rana clamitans*, *R. utricularia*, *R. catesbeiana*, and *Acris crepitans*; ringed salamander, Woodhouse's toad, green frog, southern leopard frog, bullfrog, and cricket frog), and small frogs commonly associated with wet to flooded brushy areas (*Pseudacris triseriata* and *P. crucifer*; upland chorus frog and spring peeper).

Axis two was interpreted as an elevational gradient, with higher-elevation sites with low scores on this axis, and lower-elevation sites with high scores (fig. 3). The most upland community was that of the small, generally fishless, often ephemeral ponds, which were often artificially created wildlife ponds, shallow roadside ditches, or scooped-out depressions next to a road berm and were often found near roads at the boundary ridges of the watersheds (American toad, gray treefrog, spotted salamander, central newt, and eastern narrowmouth toad). *Bufo americanus,* a member of this group, also was frequently found at dry, ridge-top sites away from ponds. Next in the ordination of axis two fell the terrestrial salamanders (*P. albagula* and *P. serratus*), who were found on terrestrial sites at all elevations and so whose centroids fell out in the center of the gradient. Species found lower down in the watersheds included those of medium-to-large ponds and flooded wetlands, habitats characteristic of lowlands (ringed salamander, Woodhouse's toad, green frog, southern leopard frog, bullfrog, cricket frog, upland chorus frog, and spring peeper). Finally, those salamanders that frequented small streams or shallow backwaters of larger water courses (*D. brimleyorum* and *E. multiplicata*) were plotted; frequently their habitat adjoined the largest streams at the bottom of the watershed.

Axis three was more difficult to interpret (and had less statistical explanatory power) (table 5), but knowledge of the species sorting out at the extremes of this presumed gradient led us to conclude that it reflected differences in canopy cover, low to high on the vertical axis (fig. 4). *Bufo woodhousei* breeds along the shores of rather large ponds and is rarely encountered in other habitats. Such sites have zero to low canopy cover. *Pseudacris triseriata* generally was found in or near water in flooded, brushy areas with sparse overstory. The central newt usually was found swimming in small, open ponds with limited canopy cover, and gray treefrog frequented ridge-top hardwoods, also with limited canopy cover. On the opposite extreme, we found the rather rare four-toed salamander (*Hemidactylium scutatum*) on dark plots with maximal canopy cover, either in closed-in pine plantations or very dense hardwoods. The centroids of the rest of the species fell out in the middle of this axis, implying that these species were limited to intermediate levels of canopy cover, or they were found on plots with both low and high canopy cover (thus, their centroids would fall out along the middle of this axis).

Scattergrams of plot scores onto the same three axes, aggregating the plots of the separate watersheds, offered additional support for our interpretation of the environmental gradients that the axes represented. Ellipses enclosing 95 percent of the plots of the more intensively managed watersheds, Little Glazypeau and North Alum, were a bit larger than those enclosing 95 percent of the plots of their less intensively managed counterparts, Bread Creek and South Alum (figs. 5 and 6). In fact, ellipses for Little Glazypeau and North Alum completely enclose those for Bread Creek and South Alum for axes three vs. one (fig. 6). In DCA, broader extent of plots along axes means that those plots offer more varied habitat (plot to plot) for more varied communities of the organisms; i.e., greater beta diversity. Thus, even though plots of the four watersheds were not strongly separated (which meant communities were not all that different), there was the suggestion that North Alum and Little Glazypeau offered a more heterogeneous, patchy habitat mosaic than Bread Creek and South Alum. The former watersheds, in addition to plots with intermediate DCA scores, had more extreme aquatic plots (axis one) and both open and closed-canopy plots (axis three) than the latter. This enhanced patchiness appeared to have furnished suitable habitat conditions for a greater diversity of amphibians; i.e., greater between-site, or beta, diversity.

CONCLUSIONS

Although watershed sizes varied more than two-fold, there was no relationship between species richness and watershed area. Likewise, there was no relationship between species richness and intensity of management. There is the suggestion, however, that species diversity and species evenness may relate to intensity of management. The two most intensively managed watersheds (Little Glazypeau and North Alum) showed overall species diversity higher

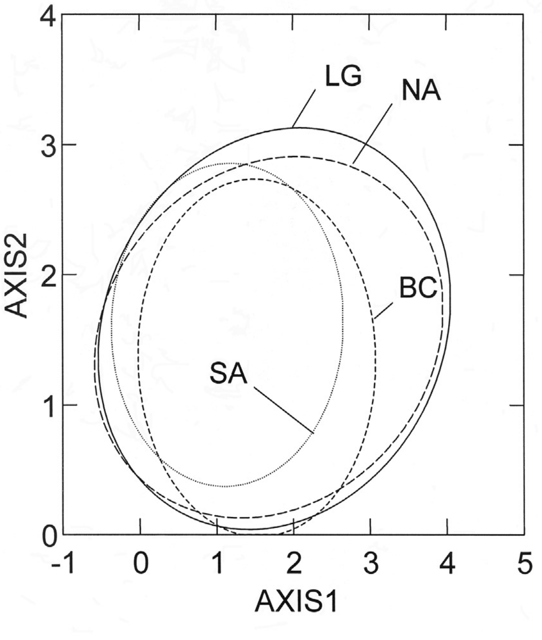

Figure 5—Ninety-five percent sample probability ellipses of the plot scores from detrended correspondence analysis (DCA) of the four watersheds: DCA axis two vs. one (SA = South Alum, BC = Bread Creek, NA = North Alum, LG = Little Glazypeau).

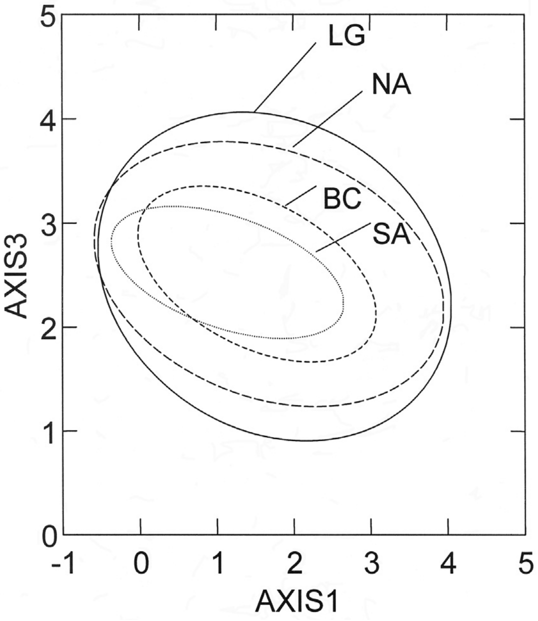

Figure 6—Ninety-five percent sample probability ellipses of the plot scores from detrended correspondence analysis (DCA) of the four watersheds: DCA axis three vs. one (SA = South Alum, BC = Bread Creek, NA = North Alum, LG = Little Glazypeau).

than that of the least intensively managed watersheds (Bread Creek and South Alum), with the diversity of North Alum distinctly higher than the rest. Recall that North Alum had large areas of relatively undisturbed middle and upper slope forests along with intensively managed Weyerhaeuser plantations (representing a diversity of successional stages) on the lower slope and valley topographic positions. Since species richness of all four watersheds was almost identical, the pattern of species diversity came about because the most intensively managed watersheds had a more even distribution of species. Cluster analysis of community similarity using Morisita's index, which incorporates Simpson's index of dominance (Simpson 1949), resulted in the same pattern. The most divergent watershed was the one least intensively managed, South Alum, and the two most intensively managed watersheds, Little Glazypeau and North Alum, grouped together. In fact, of all the pairs, only little Glazypeau and North Alum fell out statistically indistinguishable based on Morisita's index. This high degree of similarity was due primarily to the higher species evenness in both these watersheds.

At the plot level, no relationship to silviculture was strongly evident. None of our community parameters (total number, species richness, and species diversity) differed significantly among watersheds, although the least disturbed watersheds (Bread Creek and South Alum) had plots with nominally higher species richness and species diversity, unlike the tendency observed at the watershed level.

All told, the amphibian communities on the four watersheds did not appear to be that different, even though the level of similarity of most pairwise comparisons was gauged to be less than that expected by chance. However, when classifying the four watersheds into just two levels of intensity of management, there is the qualitative suggestion that whereas alpha diversity (within-plot diversity) was related negatively to intensity of forest management, beta diversity (among-plot diversity) was related positively to intensity of forest management. Both tendencies, however, were only weakly demonstrated. At face value, it would appear that whereas the individual plots on the more intensively managed watersheds harbored perhaps somewhat less diverse assemblages of amphibians, the overall diversity of the watershed was putatively higher for these than the less intensively managed watersheds. These tendencies suggest that forest management applied at a large scale in the Ouachita Mountains of Arkansas may increase overall amphibian diversity (almost completely through increased evenness), probably because such habitat manipulation increases the diversity of habitat patches (timber stands of different ages and plant species distributions). Along with an increase of diversity of habitat patches comes increased diversity of canopy, litter, understory, down wood, soil moisture, stream hydrology, and various other habitat features that impact amphibians. This suggested increased beta diversity, which was not demonstrated statistically, comes at a cost of decreased alpha diversity, however (also not demonstrated statistically).

One obvious environmental difference among the watersheds that affected beta diversity along axis one was the presence or absence of large ponds, which represented important habitat for the more aquatic frogs. South Alum lacked these large, permanent ponds; Bread Creek also lacked them but had some moderate-sized ponds; North Alum had some large ponds; and Little Glazypeau had the most, primarily because they were constructed as "heliponds" to supply water for fire control. Consequently, Bread Creek and especially South Alum lacked plots with high scores along DCA axis one (an environmental gradient from terrestrial to aquatic) and were relatively impoverished in species like bullfrog, southern leopard frog, and cricket frog. The DCA suggests roughly the same pattern of beta diversity among the watersheds as that implied from the watershed-wide Shannon-Wiener diversity indices (table 1), but neither analysis implies a strong difference among watersheds.

Our data weakly suggest that the more intensely managed watersheds held a greater diversity (greater evenness of about the same number of species) of amphibians than the less intensely managed watersheds. But such a difference may not relate to management at all. There may be other characteristics of Little Glazypeau and North Alum that favor a greater diversity of amphibians. For example, both these watersheds are larger than either Bread Creek or South Alum. North Alum is over three times larger than Bread Creek. It is well known that number of species increases with area of study plots, the familiar species-area curves of islands and mainland sites (Pianka 2000). Although we did not observe more species on the largest watershed, perhaps area alone furnished conditions that promoted higher species evenness, hence higher species diversity. The large size of North Alum means that it may have offered a greater variety of habitats that might have favored a greater variety of amphibians, and it was North Alum that held the highest overall species diversity and evenness of all four watersheds (table 1), even though it was not the most intensively managed watershed. There also may be other characteristics that relate to the amphibian communities found on the watersheds. These are just four watersheds picked to vary along a management continuum, but they also may vary in other ways. Without a suitable set of replicate watersheds representing various levels of forest management, it is impossible to randomize all these other variables and to assess the relative effect of forest management on amphibian communities.

One example from our study illustrates that concern. We found no central newts on Little Glazypeau, but we found them on the other watersheds, especially North Alum (also rather intensively managed). Little Glazypeau lacked small, permanent, fishless ponds whereas the other three watersheds had them, mostly man-made ones established for wildlife. Predatory fish and newt species often do not coexist (Beebee 1997, Hecnar and M'Closkey 1997). Both are predatory and have high prey overlap. Apparently, fish are more efficient predators and can outcompete the newts; introduced centrarchids can quickly eradicate *N. viridescens* from small ponds (Hecnar and M'Closkey 1997). Without the refuge of small, permanent, fishless ponds in which *N. viridescens* breeds, Little Glazypeau may not support this species, and the lack of these ponds may have nothing to do with commercial forest management. In fact, most of the small ponds were established artificially in the other watersheds for the benefit of wildlife, but this was not done at Little Glazypeau. In contrast, permanent heliponds established by Weyerhaeuser in Little Glazypeau were occasionally stocked with fish by local residents or fishermen, who released catches from nearby lakes or ponds for recreational fishing.

MANAGEMENT IMPLICATIONS

Our study qualitatively suggests that intensive forest management may decrease local, plot-wise (alpha) amphibian species diversity, but in turn may increase overall, watershed (beta) amphibian species diversity, perhaps by increasing the diversity of available habitats. However, neither tendency was statistically demonstrated because plots were not distinctly different among watersheds, and overall amphibian communities of the four watersheds were extremely similar. Taken together, our data suggest that intensive silviculture as practiced in the Ouachita Mountains of west-central Arkansas is not detrimental to landscape-level amphibian communities. This is probably true because even under the most intensive forest management, stand sizes are large, riparian zones are largely left intact, and ponds are created either for the benefit of wildlife or for a water supply for fire control. It is important to maintain those practices to conserve and maintain existing amphibian diversity. We recommend that land managers construct and/or maintain both large and small ponds, critical breeding habitat for many species of amphibians. Especially small, often ephemeral, vernal ponds are important for amphibians (Dodd and Cade 1998, Semlitsch and others 1996, Semlitsch and Bodie 1998). Because fish are important predators of amphibian eggs and larvae (Bradford 1989, Denton and Beebee 1991, Grubb 1972, Webb and Joss 1997) and are known to completely eliminate some species from permanent ponds (Fisher and Shaffer 1996, Hecnar and M'Closkey 1997), some attempt should be made to maintain some small, permanent, fishless ponds on managed watersheds. In addition to mostly terrestrial species and those characteristic of small lowland steams and backwaters, we found two large amphibian communities associated either with small, often ephemeral, often upland ponds, or large, permanent, more lowland ponds. Presence and maintenance of these two classes of ponds should be part of forest-management practices in the Ouachita Mountains.

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REPTILE COMMUNITIES UNDER DIVERSE FOREST MANAGEMENT IN THE OUACHITA MOUNTAINS, ARKANSAS

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Joseph P. Phelps, and David M. Leslie, Jr.**[1]

Abstract—From May 1995 to March 1999, we censused reptiles in the Ouachita Mountains, Arkansas, on approximately 60 plots on each of four forested watersheds five times per year, with new plots each year. We found that the least intensively managed watershed had significantly lower per-plot reptile abundances, species richness, and diversity. Despite these differences, community similarities were high (0.89-0.98, Morisita's index) between all watersheds. The least intensively managed watershed had nominally higher overall species evenness and diversity. Further inspection revealed that this was due to high dominance in the more intensively managed watersheds by two species that were not~~as~~ as dominant in the least intensively managed watershed. Detrended correspondence analysis revealed communities separating out on the basis of presumed gradients of canopy cover, terrestrial-aquatic, and a complex gradient of humidity/soil moisture.

INTRODUCTION

Recently, it has been reported that many reptile populations are experiencing global declines (Gibbons and others, 2000) akin to, if not more serious than, the highly publicized global declines in amphibian populations (Blaustein and Wake 1995, Houlahan and others 2000, Phillips 1990, Wake 1991). This realization has led to a call for increased long-term research on these often ignored taxa.

Reptiles, like amphibians, commonly experience natural population fluctuations and extinctions (Blaustein and others, 1994; Pechmann and others, 1991). However, not all observed declines can be categorized as natural (Gibbons and others, 2000). Possible causes for these declines include habitat loss and degradation, climatological change, introduction of exotic species, environmental pollution, disease and parasites, and unsustainable harvesting.

Loss of suitable habitat is considered by some scientists to be the largest single factor responsible for declines in amphibian populations (Alford and Richards, 1999). Likewise, it has been suggested that habitat loss due to urbanization, agriculture, and silviculture may play an important role in declines of reptile populations (Gibbons and others, 2000). Despite the implication of silvicultural practices in these declines, little is known about the habitat parameters that influence reptile communities, and even less is known about landscape-level environmental influences.

We report results of pretreatment data collection for a large-scale, long-term, field study of reptile communities and the influence of habitat and landscape environmental variables in four watersheds of the Ouachita Mountains of west-central Arkansas. Following the pretreatment stage, sections of the watersheds will be subjected to different forest management to achieve a variety of specific "desired future conditions." After treatment, data on reptile communities will again be collected and used to quantify community changes and to compare with the predictions of multivariate community models that we are developing. This long-term study is one component of Phase III of the Ouachita Mountains Ecosystem Management Research Project; the wildlife component of this cooperative effort involves Weyerhaeuser Company, National Council of the Paper Industry for Air and Stream Improvement, Oklahoma State University, Oklahoma Cooperative Fish and Wildlife Research Unit, University of Arkansas Monticello, Ouachita National Forest, and Southern Research Station of the USDA Forest Service.

The objectives of our overall study are to: (1) characterize reptile and amphibian communities in four watersheds representing markedly different forest-management strategies in the Ouachita Mountains, Arkansas; (2) develop and validate models for predicting community composition based on site, stand, and landscape parameters; and (3) develop recommendations to promote maintenance of reptile and amphibian communities in managed forest landscapes. This report contains results for reptiles of the pretreatment data analysis performed at the end of four survey years.

METHODS

Study Areas

The study was conducted on four 1500- to 4000-ha watersheds under different intensities of management in Garland and Saline counties near Hot Springs, Arkansas. The watersheds differed markedly with respect to factors such as mean rotation lengths, forest type diversity, stand sizes and ages, and the amount of natural second-growth coverage (Guldin and others, in press; Tappe and others, in press).

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Little Glazypeau, a watershed located some 22 km southwest of the other three watersheds (that were contiguous) and managed largely for sawlog production by Weyerhaeuser Company, represented our most intensively managed watershed. Much of the second-growth shortleaf pine (*Pinus echinata*)-hardwood forest that originally covered this watershed had been harvested and planted to loblolly pine (*P. taeda*) plantations of 9-142 ha. Typically, these plantations were thinned twice, pruned to 5-8 m high, fertilized, and harvested at 30-35 years old. The remaining, selectively-harvested acreage in the watershed occured on rocky ridgetops, steep slopes, and streamside management zones that were retained for watershed protection and to provide habitat diversity for wildlife. South Alum, an experimental section of the Ouachita National Forest that has received minimal logging for > 80 years, represented the least intense level of silviculture. South Alum was almost entirely USDA Forest Service land and consisted of mature forest over most of the area. Bread Creek and North Alum fell in between these extremes of forest management. From independent records of forest management and present composition of number, age, and distribution of pine (mostly *Pinus echinata*) plantations, Bread Creek was considered less intensely managed than North Alum (Tappe and others, in press). Bread Creek was primarily USDA Forest Service land and had been managed according to prevailing Forest Service standards and guidelines for several decades, whereas North Alum was of mixed ownership, with about half of the area under more intensive Weyerhaeuser Company management (comparable to Little Glazypeau) and half under USDA Forest Service management. North Alum displayed characteristics of a diverse range of management activities, ranging from no management along steeper slopes and higher elevations to intensively-managed pine plantations, mainly at lower elevations. Thus, the watersheds, in order of intensity of forest management (from least to most), were South Alum, Bread Creek, North Alum, and Little Glazypeau. These same watersheds, in order of size, were South Alum (1500 ha), Bread Creek (1535 ha), Little Glazypeau (2273 ha), and North Alum (3961 ha).

Sampling Plots

We surveyed reptiles and amphibians using area-constrained searches on a subset (56) of the 75-235 plots per watershed that were established each year for breeding bird surveys (Tappe and others, in press). Bird plots were established at 200-m intervals along >100 km of parallel transects (oriented approximately north-south across prevailing topography) that were established in 1995 over the 4 watersheds. These same transects were used in 1996, but new plots were established by shifting plot centers 100 m. In 1997, approximately 110 km of new transects were established between those of 1995; new plots were established in 1998 along these new transects as in 1996. The subset of plots that were used for reptile surveys were selected to represent a cross-section of slopes, aspects, forest types, stand conditions, and aquatic habitats. The center of our 20-m-radius (0.13-ha) plots also was the center of a bird sampling plot. Plastic flagging was used to delineate plot boundaries on all reptile plots. In each watershed, we selected reptile plots to ensure there were at least 12-15 of them in aquatic habitats, which consisted of springs, streams, and man-made ponds that had been established to benefit wildlife (Forest Service wildlife ponds) and/or as sources of water for fire fighting. An additional four plots per watershed per year were established off the transects at these ponds (or at wide pools in streams at the bottom of a watershed) to ensure that we had equivalent sampling effort at these aquatic habitats. Plots at these aquatic sites were established so that roughly half of the plot was water and half was land. Thus, we surveyed 60 plots per year per watershed.

Reptile Surveys

Trained crews of 3-5 individuals surveyed each set of 60 plots per watershed during daylight hours 5 times a year from May 1995 to March 1999: early May, late May, mid-June, early October, and early the following March. Plots were surveyed entirely by visually searching vegetation and the ground surface, and by lifting cover objects (rocks, logs, and debris); the latter were replaced to minimize impacts on subsequent surveys. Reptiles that were seen and identified were tallied; those that required capture for identification were released at the point of capture before leaving the plot.

Data Analysis

Reptile count data were pooled across the five sampling periods per year, but data from each plot served as separate samples. Data collected in the first year (1995) from South Alum and North Alum were excluded from analyses due to differences in sampling effort by former collaborators, and data from a few other plots were discarded when five surveys per year were not attained. Thus, analyses presented here are based on data from 833 plots, each censused five times, for a total of 4,165 censuses. Watershed- and plot-level data were analyzed differently as described in the following two paragraphs.

Watershed level—The following community indices were first calculated at the watershed level: number of individuals by species for all years combined, reptile abundance, species richness, (beta) species diversity [H'; Shannon-Wiener diversity index (Shannon and Weaver 1949)], and species evenness [diversity divided by maximal diversity, or H'/(ln number of species); Pielou 1966]. With only four watersheds to compare, no statistical tests were employed. Because the Shannon-Wiener diversity index is quite sensitive to sample size and because we did not have an equal effort among all watersheds (two watersheds with data from only three years; see above), we randomly reordered the plots within each watershed, pooling years, and plotted the Shannon-Wiener diversity index of each watershed for cumulative sets of plots up to the total number of plots for each watershed. Such a plot would show if diversity approached an asymptote as cumulative plots increased and if our total number of plots per watershed was sufficient to adequately estimate species diversity.

We computed Morisita's index of community similarity (Morisita 1959) between all pairs of watersheds, pooling data for the entire study for each watershed. This index is desirable because sample sizes and species diversities of the communities being compared have little influence on its calculation (Morisita 1959; Wolda 1981). To statistically compare the various Morisita's indices of community similarity, we conducted randomization tests (Biondini and others 1988). For each pair of watersheds, we randomly reassigned plots between them (retaining the sample size of each watershed) and computed Morisita's index for these two "synthetic" communities. We repeated this procedure 1000 times and tabulated the number of times the recomputed index was smaller than or equal to the "actual" index. If less than 100 recomputed indices fell below or equal to the "actual" index, then those two watersheds were considered different ($p < 0.10$) in species composition.

Plot level—For analyses at the plot level, we used mixed model, two-way ANOVAs (ANOVAs with both random and fixed effects; PROC MIXED, SAS 1999) to test for differences among watersheds, years, and year * watershed interaction. We recognize that our sample of plots drawn from each of four watersheds is pseudoreplicated (Hurlbert, 1984), but the large scale of this study prevented sampling of replicate watersheds for each treatment. While results of our ANOVA must be interpreted with caution due to this pseudoreplication, we feel that the analysis nevertheless suggests likely ecological patterns that deserve attention. The response variables of the ANOVAs were (1) reptile abundance per plot, (2) species richness per plot, and (alpha) plot diversity (Shannon-Wiener index). For count variables (1 and 2) we used a square root transformation, SQR (count + 0.5). Even with those transformations, our data did not fully meet assumptions of normality and homogeneity of variances, but the Satterthwaite algorithm of the mixed model ANOVA is relatively robust to abuses of these assumptions, especially of homogeneity of variances (SAS 1999), and so we proceeded with these parametric analyses. We recognized differences in weather between years, not of interest to us here, and included year effects and year * watershed interaction as random effects, not to be statistically interpreted. The fixed factor (watersheds) was tested for statistical significance at $p < 0.10$. If a significant watershed effect was found, we used LSD to evaluate pairwise differences between any two watersheds.

Detrended canonical correspondence analysis (DCA)—To appraise reptile community composition, develop preliminary hypotheses of presumptive environmental gradients influencing these communities, and compare graphically the environments and reptile communities of the four watersheds, we used DCA (ter Braak and Prentice 1988, ter Braak and Šmilauer 1998), pooling plots from all four watersheds and years. Application of DCA to our data allowed for a more detailed inspection of reptile communities at the plot level and how they were distributed along inferred environmental gradients. DCA is probably the most widely employed eigenanalysis-based ordination technique used by community ecologists. It is an indirect ordination method that orders plots with similar compositions of species along multiple axes simultaneously. The statistical algorithm is to calculate sample scores of each plot as a weighted average of the species scores, and species scores as a weighted average of samples scores; iterations are repeated until there is no further change in scores, at which time samples (plots) with similar animal communities appear clustered when plotted on multiple axes. Environmental gradients are inferred from the pattern of species and/or plots and the biologist's knowledge of the species. As a step beyond single-number summary statistics like diversity indices or Morisita's indices, DCA results in a cloud of points for separate species in n-dimensional space, conventionally viewed as centroids (averaged central tendency) in two dimensions at a time. In other words, DCA results in a pattern, not a number. For our analysis, rare species (less than three individuals encountered over all four watersheds for all four years) and plots where no reptile species were found were excluded due to computational constraints.

RESULTS AND DISCUSSION

We found 1,877 individuals of 35 reptile species during our four-year study (table 1). Total species diversity was 2.38 for all watersheds pooled. We found a mean of 2.25 reptiles per plot (median = 1.00, range = 0-22).

Watershed Level

Despite large differences in size of watersheds and substantial differences in management intensities, the reptile communities of these four watersheds were fairly similar. Species richness ranged from 26-28 species per watershed, and 19 of the total set of 35 species (54 percent) were found on all four watersheds (table 1). The watersheds differed some in both overall species diversity and evenness; the order from lowest to highest by diversity was Bread Creek, Little Glazypeau, North Alum, and South Alum (table 1). For evenness, the order from lowest to highest was similarly polarized as Bread Creek, North Alum, Little Glazypeau, and South Alum. The middle watersheds in both of these rankings, North Alum and Little Glazypeau, were virtually identical in these measures. Recalculated diversity indices against cumulative plots showed that diversity leveled off after about 40-100 plots (fig. 1), well below the lowest total of plots for any watershed. South Alum showed distinctly higher overall diversity than the other watersheds. Bread Creek had the lowest overall diversity and North Alum and Little Glazypeau were intermediate and nearly indistinguishible from each other.

The dominant species found in all watersheds were the ground skink *(Scincella lateralis)* and the western fence lizard (*Sceloporus undulatus*), representing on average 51.1 percent of total reptiles encountered in each watershed community. Additionally, species composition of the six most ubiquitous species of each watershed was strikingly similar (table 2).

We examined the overall set of species to see if there were any species absent from all but one watershed, or present in only one watershed. The flathead snake *(Tantilla gracilis)* was absent from Little Glazypeau; the six-line racerunner (*Cnemidophorus sexlineatus)* was not found on any Bread Creek plots; and the green anole (*Anolis carolinensis)* and the speckled kingsnake (*Lampropeltis getula)* were not found in South Alum watershed. The western diamondback rattlesnake *(Crotalus atrox)* was found only in Little Glazypeau; the Great Plains ratsnake *(Elaphe guttata)*, eastern coachwhip (*Masticophis flagellum)*, diamondback water snake (*Nerodia rhombifer)*, and the rough earth snake (*Virginia striatula)* were recorded only in North Alum; the scarlet snake (*Cemophora coccinea)* was found only in

Table 1—Reptile abundance on four watersheds in the Ouachita Mountains, Arkansas, 1995-1999

Species	Little Glazypeau	North Alum	Bread Creek	South Alum
Agkistrodon contortrix (copperhead)	16	7	8	16
Agkistrodon piscivorus (cottonmouth)	4	20	7	10
Anolis carolinensis (green anole)	43	4	18	0
Carphophis vermis (western worm snake)	11	9	29	13
Cemophora coccinea (scarlet snake)	0	0	1	0
Cnemidophorus sexlineatus (six-line racerunner)	2	7	0	1
Coluber constrictor (black racer)	8	11	7	1
Crotalus atrox (western diamondback rattlesnake)	1	0	0	0
Crotalus horridus (timber rattlesnake)	1	0	1	0
Diadophis punctatus (western ringneck snake)	35	18	44	29
Elaphe guttata (Great Plains ratsnake)	0	1	0	0
Elaphe obsoleta (black ratsnake)	3	1	1	1
Eumeces anthracinus (coal skink)	21	12	26	11
Eumeces fasciatus (five-line skink)	48	22	46	16
Eumeces laticeps (broadhead skink)	10	4	11	3
Heterodon platirhinos (eastern hognose)	0	0	1	1
Lampropeltis getula (speckled kingsnake)	5	3	1	0
Lampropeltis triangulum (milksnake)	1	1	4	2
Masticophis flagellum (eastern coachwhip)	0	1	0	0
Nerodia erythrogaster (yellowbelly watersnake)	2	8	1	2
Nerodia rhombifer (diamondback watersnake)	0	1	0	0
Nerodia sipedon (midland watersnake)	1	0	0	4
Opheodrys aestivus (rough green snake)	3	9	2	5
Scincella lateralis (ground skink)	187	111	160	69
Sceloporus undulatus (northern fence lizard)	93	110	209	47
Sistrurus miliarius (western pigmy rattlesnake)	2	4	1	3
Storeria dekayi (brown snake)	6	1	2	2
Storeria occipitomaculata (northern redbelly snake)	12	5	7	10

continued

Table 1—Reptile abundance on four watersheds in the Ouachita Mountains, Arkansas, 1995-1999 (continued)

Species	Little Glazypeau	North Alum	Bread Creek	South Alum
Tantilla gracilis (flathead snake)	0	3	4	1
Terrapene carolina (eastern three-toe box turtle)	46	20	16	26
Terrapene ornata *(ornate box turtle)*	7	0	0	1
Thamnophis proximus (western ribbon snake)	0	0	0	1
Thamnophis sirtalis (red-sided garter snake)	3	2	7	2
Virginia striatula (rough earth snake)	0	1	0	0
Virginia valeriae (smooth earth snake)	4	4	6	5
Total	575	400	620	282
Species richness	27	28	26	26
Species diversity	2.34	2.35	2.12	2.51
Species evenness	0.87	0.86	0.84	0.92

Data are arrayed (left to right) from the most to the least intensively managed watersheds.

Bread Creek; and the western ribbon snake (*Thamnophis proximus)* was found only in South Alum watershed.

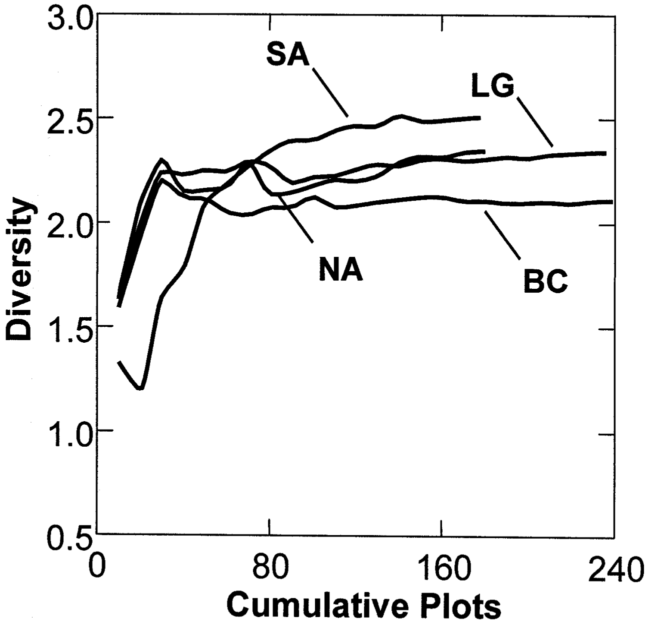

Figure 1—Diversity of each watershed against cumulative number of plots (in random order) included in recalculation. SA = South Alum; BC = Bread Creek; NA = North Alum; LG = Little Glazypeau.

Based on Morisita's index of community similarity, watershed reptile communities were highly similar, ranging from 0.89 to 0.98 (a value of 1.00 means identical communities; table 3). Bread Creek differed the most from the other watersheds, displaying an index of 0.89 with both Little Glazypeau and South Alum. Despite generally high indices of community similarity, all indices [except that between North Alum and Bread Creek (O.98)] were statistically significant by the randomization tests; i.e., all watershed pairs except this one were significantly different from each other beyond that expected by chance.

Plot Level

We found a significant watershed effect for all three plot-level measures (table 4): reptile abundance ($F_3 = 2.55$, $p = 0.05$), richness ($F_3 = 2.53$, $p = 0.06$), and diversity ($F_3 = 2.64$, $p = 0.05$). South Alum had plots with significantly fewer reptiles, lower species richness, and lower species diversity than each of the other watersheds based on ANOVA (table 4). None of the pairwise comparisons for the other watersheds were significant ($p > 0.10$).

Detrended Correspondence Analysis

Detrended correspondence analysis indicated relatively long environmental gradients along the first three axes for reptile communities in the four watersheds (table 5). The fourth and additional axes contributed little to the pattern of

Table 2—Six most common reptile species found in each watershed in order of decreasing abundance

Little Glazypeau	n	North Alum	n	Bread Creek	n	South Alum	n
Scincella lateralis (ground skink)	187	*Schincella lateralis* (ground skink)	111	*Sceloporus undulatus* (northern fence lizard)	209	*Scincella lateralis* (ground skink)	69
Sceloporus undulatus (northern fence lizard)	93	*Sceloporus undulatus* (northern fence lizard)	110	*Scincella lateralis* (ground skink)	160	*Sceloporus undulatus* (northern fence lizard)	47
Eumeces fasciatus (five-line skink)	48	*Eumeces fasciatus* (five-line skink)	22	*Eumeces fasciatus* (western ringneck snake)	46	*Diadophis punctatus* five-line skink)	29
Terrapene carolina eastern three-toe box turtle)	46	*Terrapene carolina* (eastern three-toe box turtle)	20	*Diadophis punctatus* (western ringneck snake	44	*Terrapene carolina* (eastern three-toe box turtle)	26
Anolis carolinensis (green anole)	43	*Agkistrodon piscivorus* (cottonmouth)	20	*Carphophis vermis* (western worm snake)	29	*Eumeces fasciatus* (five-line skink)	16
Diadophis punctatus (western ringneck snake)	35	*Diadophis punctatus* (western ringneck snake)	18	*Eumeces anthracinus* (coal skink)	26	*Carphophis vermis* (western worm snake)	13

Table 3—Morisita's index of community similarity for all pairs of watersheds

Watershed	Little Glazypeau	North Alum	Bread Creek	South Alum
Little Glazypeau	1.00			
North Alum	0.93[a]	1.00		
Bread Creek	0.89[a]	0.98	1.00	
South Alum	0.96[a]	0.94[a]	0.89[a]	1.00

Index ranges from 0 to 1, where 1 means communities are identical.

[a] Significantly dissimilar by Randomization Test, $p < 0.10$.

Table 4—Number of plots surveyed (1995–1999), mean reptile abundance per plot, mean species richness per plot, and mean species diversity per plot by watershed

Watershed	n	Mean abundance[a]	Mean species richness[b]	Mean species diversity[c]
Little Glazypeau	236	2.44	1.58	0.408
North Alum	180	2.22	1.54	0.404
Bread Creek	239	2.59	1.69	0.453
South Alum	178	1.58	1.23	0.292

Vertical bars connect those watersheds not statistically different as indicated by post-hoc pairwise contrasts.

[a] ANOVA: $F_3 = 2.55$, $p = 0.05$.

[b] ANOVA: $F_3 = 2.53$, $p = 0.06$.

[c] ANOVA: $F_3 = 2.64$, $p = 0.05$.

Table 5—Eigenvalues of first 4 axes of detrended correspondence analysis of 25 reptile species distributed among a pooled total of 403 plots on the 4 watersheds[a]

Axis	Eigenvalue
One	0.513
Two	0.441
Three	0.334
Four	0.287

[a] An eigenvalue is the correlation coefficient between the plot scores and species scores along a given axis where each axis is orthogonal [independent] to all previous axes in the analysis.

community organization because each additional axis explains only residual variation not already incorporated into the DCA. In other words, plots and species scores were relatively tightly correlated with each other along an appreciable stretch of at least the first three axes (eigenvalues range from 0 to 1: high eigenvalues meant that clouds of points were spread linearly along each axis, and low eigenvalues meant that points were clustered at the center of each axis). Species' centroids plotted against axes two vs. one (fig. 2) and against axes three vs. one (fig. 3) showed strong separation of species.

The pattern of species' centroids along axis one suggested that it was a measure of low-to-high canopy cover, left to right (fig. 2). Most of the species scoring lowest on this axis were typically found in more open or edge habitats. These species were *Tantilla gracilis* (flathead snake), *Sceloporus undulatus* (northern fence lizard), *Nerodia erythrogaster* (yellowbelly watersnake), *Elaphe obsoleta* (black ratsnake), and *Cnemidophorous sexlineatus* (six-line racerunner). To

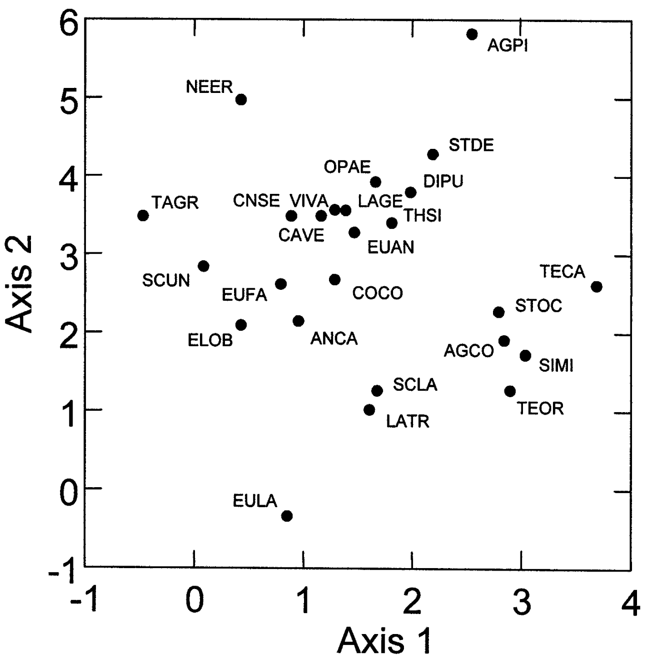

Figure 2—Centroids of species' scores from detrended correspondence analysis (DCA) of censused plots on all four watersheds pooled: DCA axis two vs. one. AGCO = *Agkistrodon contortrix*; AGPI = *A. piscivorous*; ANCA = *Anolis carolinensis*; CAVE = *Carphophis vermis*; CNSE = *Cnemidophorus sexlineatus*; COCO = *Coluber constrictor;* DIPU = *Diadophis punctatus*; ELOB = *Elaphe obsolete*; EUAN = *Eumeces anthracinus*; EUFA = *E. fasciatus*; EULA = *E. laticeps*; LAGE = *Lampropeltis getula*; LATR = *L. triangulum*; NEER = *Nerodia erythrogaster*; OPAE = *Opheodrys aestivus*; SCLA = *Scincella lateralis*; SCUN = *Sceloporus undulates*; SIMI = *Sistrurus miliarius*; STDE = *Storeria dekayi*; STOC = *S. occipitomaculata*; TAGR = *Tantilla gracilis*; TECA = *Terrapene Carolina*; TEOR = *T. ornate*; THSI = *Thamnophis sirtalis*; VIVA = *Virginia valeriae*.

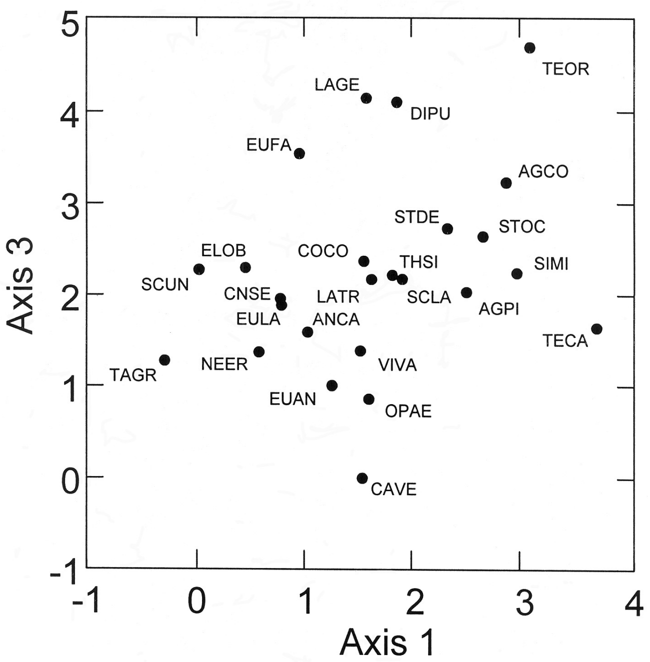

Figure 3—Centroids of species' scores from detrended correspondence analysis (DCA) of censused plots on all four watersheds pooled: DCA axis three vs. one. AGCO = *Agkistrodon contortrix*; AGPI = *A. piscivorous*; ANCA = *Anolis carolinensis*; CAVE = *Carphophis vermis*; CNSE = *Cnemidophorus sexlineatus*; DIPU = *Diadophis punctatus*; ELOB = *Elaphe obsolete*; EUAN = *Eumeces anthracinus*; EUFA = *E. fasciatus*; EULA = *E. laticeps*; LAGE = *Lampropeltis getula*; LATR = *L. triangulum*; NEER = *Nerodia erythrogaster*, OPAE = *Opheodrys aestivus*; SCLA = *Scincella lateralis*; SCUN = *Sceloporus undulates*; SIMI = *Sistrurus miliarius*; STDE = *Storeria dekayi*; STOC = *S. occipitomaculata*; TAGR = *Tantilla gracilis*; TECA = *Terrapene Carolina*; TEOR = *T. ornate*; THSI = *Thamnophis sirtalis*; VIVA = *Virginia valeriae*.

the far right, scoring highest on axis one, were mostly species typically associated with closed-canopy forested habitats, like *Terrapene carolina* (eastern three-toe box turtle), *Sistrurus miliarius* (western pygmy rattlesnake), *Storeria occipitomaculata* (northern redbelly snake), and A*gkistrodon contortrix* (copperhead). Species found at the middle of this axis were found in sites with intermediate or mixed canopy cover.

Axis two was interpreted as an aquatic gradient, with terrestrial sites having low scores on this axis and aquatic sites having high scores (fig. 2). The most terrestrial species (the lowest scores on axis two) included *Eumeces laticeps* (broadhead skink), *Lampropeltis triangulum* (milksnake), *Terrapene ornata* (ornate box turtle), and *Scincella lateralis* (ground skink). Scoring highest along axis two were the aquatic species, *Agkistrodon piscivorus* (cottonmouth) and *Nerodia erythrogaster* (yellowbelly watersnake).

Axis three was more difficult to interpret (and had less statistical explanatory power, table 5), but knowledge of the species displayed at the extremes of this presumed gradient led us to conclude that it reflected a complex gradient of humidity/soil moisture (fig. 3). Species located highest along axis three are typical of more western, arid habitats of the United States. These species are *Terrapene ornata* (ornate box turtle), *Lampropeltis getula* (speckled kingsnake), and *Diadophis punctatus* (ringneck snake). Species found lowest on the axis were associated with moist substrates or humid environments: *Carphophis vermis* (western worm snake), *Ophyeodrys aestivus* (rough green snake), and *Eumeces anthracinus* (coal skink).

Scattergrams of plot scores onto the same three axes, aggregating the plots of the separate watersheds, illustrated the overall similarity of the watersheds. Ellipses enclosing 95 percent of the plots of the watersheds overlapped considerably (figs. 4 and 5). In DCA, broader extent of plots along axes means that those plots offer more varied habitat (plot to plot) in which live more varied communities of the organisms studied, i.e., greater beta diversity. Except perhaps for Bread Creek, the watersheds had comparable reaches along each axis. The reptile communities of these four watersheds were so similar that drawing conclusions about minor differences among watersheds is not reasonable.

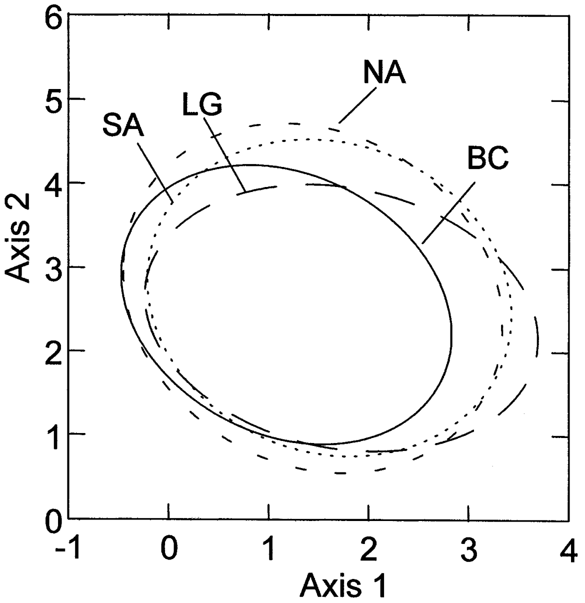

Figure 4—Ninety-five percent sample probability ellipses of the plot scores from detrended correspondence analysis (DCA) of the four watersheds: DCA axis two vs. one. LG = Little Glazypeau; NA = North Alum; BC = Bread Creek; SA = South Alum.

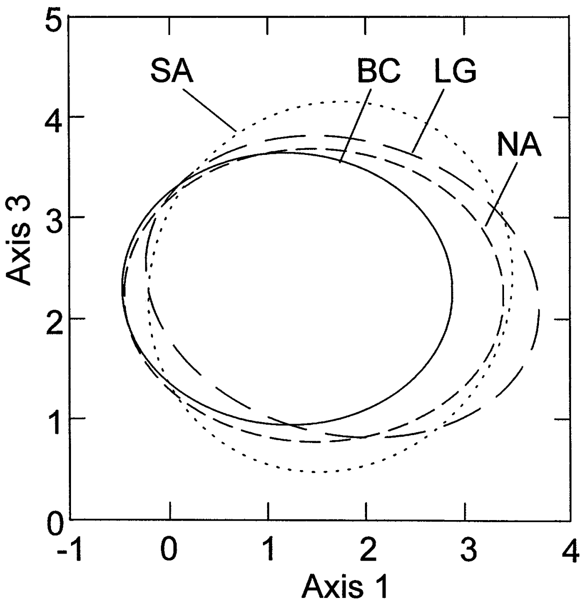

Figure 5—Ninety-five percent sample probability ellipses of the plot scores from detrended correspondence analysis (DCA) of the four watersheds: DCA axis three vs. one. LG = Little Glazypeau; NA = North Alum; BC = Bread Creek; SA = South Alum.

CONCLUSIONS

At the watershed level, total species richness per watershed varied little, 26-28 species, and most of these species were common to all watersheds, as indicated by Morisita's comparisons. Also, species diversity per watershed (beta diversity) differed little. The least intensively managed watershed, South Alum, had the nominally highest beta diversity. However, this is not necessarily due to a lack of species in the other watersheds, but instead is likely due to the dominance in the other watersheds of two species, *Scincella lateralis* (ground skink) and *Sceloporus undulatus* (northern fence lizard). The relative lack of dominance by these two species in South Alum results in greater evenness there. Because the Shannon-Wiener index incorporates evenness for its calculation, South Alum appears to be most diverse. Thus, if we reduce the recorded abundances of *S. lateralis* and *S. undulatus* in Little Glazypeau, North Alum, and Bread Creek to those found in South Alum, (69 and 47 animals, respectively) and then recalculate Shannon-Wiener diversity indices for these three watersheds, we find that our recalculated diversity index for each watershed exceeds that of South Alum (2.51) in all cases (Little Glazypeau = 2.65, North Alum = 2.67, Bread Creek = 2.59). In other words, these other watersheds possessed an inherent diversity that was higher than South Alum, plus more individuals of two reptile species.

It is well known that number of species increases with area of study plots; i.e., the familiar species-area curves of islands and mainland sites (Pianka 2000). This should relate to differences in beta diversity of entire watersheds. Little Glazypeau and North Alum were much larger than Bread Creek and South Alum. We expected, therefore, that the smaller watersheds should have had lower beta diversity. Our recalculated diversity indices adjusted for species dominance showed just this—elevated overall species diversity on the larger watersheds.

At the plot level, our data suggested that the less intensively managed watershed, South Alum, had plots with lower reptile abundance, fewer species, and decreased alpha diversity than the more intensively managed watersheds. But one must exercise some caution here; such differences may not relate to management at all. There may be other characteristics of Little Glazypeau, North Alum, and Bread Creek that affect species diversity of reptiles at the plot level. These are just four watersheds picked to vary along a management continuum, but they also may vary in other ways. Without a suitable set of replicate watersheds representing various levels of forest management, it is impossible to randomize all these other variables and to assess the relative effect of forest management on reptile communities. Nevertheless, the plot-level community differences may well be due to land management. South Alum, in contrast with the other watersheds, had virtually no logging for over 80 years and differed by having more mature, larger trees with intermediate canopy, and drastically less herbaceous vegetation, shrubs, and vines. Visually, South Alum appeared park-like. These differences made for structurally less complex within-plot habitats, possibly leading to the lower observed reptile numbers, species richness, and species diversity (alpha diversity).

MANAGEMENT IMPLICATIONS

Our study suggests that minimal forest management may result in lower local reptile abundances, species richness, and plot-wise (alpha) species diversity compared with more intensive management. However, overall watershed (beta) reptile species diversity differs little between watersheds of different management intensities. Overall reptile communities of the four watersheds were extremely similar. Taken together, our data suggest that intensive silviculture as practiced in the Ouachita Mountains of west-central Arkansas is not detrimental to landscape-level reptile communities and in fact, may be beneficial. This is probably true because even under the most intensive forest management, stand sizes are large, riparian zones are largely left intact, and ponds are created either for the benefit of wildlife or for a water supply for fire control. It is important to maintain those practices to conserve and maintain existing reptile diversity.

A possible concern with the suggestion that intensive forest management might benefit reptile communities is that rare or extremely habitat-sensitive species might suffer under this strategy. Nonetheless, we found no evidence of rare or sensitive reptile species that were being harmed by the more intensive land management. In fact, the one really rare species, C*rotalus atrox* (western diamondback rattlesnake), was found only on the most intensively managed watershed, Little Glazypeau. These findings must be tempered, though, by the fact that the Ouachita Mountains have been logged since the arrival of Europeans to the area, and because no baseline data exist for reptile communities prior to this, we have only the current diversity of all watersheds combined (gamma diversity) from our study to evaluate how rare or sensitive species might be affected by current forest management practices. Long-term monitoring is the only way to identify population trends of rare or sensitive species.

Our data suggest that diverse reptile communities in the Ouachita Mountains of west-central Arkansas would best be maintained by management decisions that create watershed-level landscape conditions with a diversity of canopy cover conditions and aquatic habitats. These two factors can be influenced and controlled by forest managers. Variation in humidity and soil moisture, also important for diverse reptile communities, may also vary with canopy cover and aquatic habitats, but these parameters might be less influenced by management decisions and more determined by the inherent physiography of a watershed (factors such as soil type, slope, aspect, elevation, and naturally-occurring aquatic habitats). We will use our data on reptile communities, plus available habitat data, to develop and validate quantitative models for predicting reptile community composition from habitat and landscape parameters.

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AQUATIC TURTLES OF DIVERSELY MANAGED WATERSHEDS IN THE OUACHITA MOUNTAINS, ARKANSAS

Joseph P. Phelps[1]

Abstract—Aquatic turtles were trapped using hoop nets in creeks and ponds located in four Ouachita Mountain watersheds (Little Glazypeau, North Alum, Bread, and South Alum Creeks). These watersheds range in management from one dominated by industrial loblolly pine (*Pinus taeda* L.) plantations to one having virtually no management for many decades. Trapping effort consisted of 212 trapnights (192 in streams, 20 in ponds) during July and August 1995 and 1996. There were 63 captures for a success rate of 0.297 captures per trap-night. Captured turtles were individually marked and released. Stream characteristics potentially related to turtle habitat were evaluated at each trapping site. Common snapping turtles (*Chelydra serpentina* L.) were the most common species captured, occurring in all watersheds and in both streams and ponds. Five other species were captured, notably the razorback musk turtle (*Sternotherus carinatus* Gray), not previously known to occur in Saline County, and the alligator snapping turtle (*Macroclemys temminckii* Harlan), an Arkansas protected species. Both species richness and number of captures were highest in the more heavily managed watersheds (Little Glazypeau and North Alum). These are also the largest creeks, so effects of management are obscured. The presence of fire-fighting ponds in these watersheds did increase richness. Excluding recaptures, number of turtles captured in streams was positively correlated ($p = 0.0059$) with an index of pool size at the trap site.

INTRODUCTION

Wildlife habitat quality may be influenced by processes beyond the scale of the traditional stand-level study. The Phase III watershed research described in many of the papers in this volume is an attempt to document the impact of forest management on plant and animal communities at a landscape scale. The four Phase III watersheds were chosen to represent a range of management intensities from industrial forestry at one extreme to a virtually unmanaged forested watershed at the other extreme.

The Little Glazypeau Creek watershed (LG) is 1740 ha in size and is the most intensively managed of the four watersheds. LG consists largely of loblolly pine (*Pinus taeda* L.) plantations less than 35 years old, interspersed with natural second growth pine-hardwood stands on ridges and near large streams. LG is owned and managed primarily by Weyerhaeuser Company. Plantations are harvested by clearcutting and regenerated by planting of seedlings. The North Alum Creek watershed (NA) is 3800 ha, and is owned and managed by Weyerhaeuser Company and the Ouachita National Forest (ONF) in about equal proportions. NA is managed operationally by Weyerhaeuser and ONF, so its management is like that described for LG and Bread Creek, respectively. The Bread Creek watershed (BC) is 1255 ha and is owned and managed primarily by ONF. BC is managed operationally for multiple uses. The ONF management regime is less intensive than Weyerhaeuser management, with longer rotations, natural regeneration of shortleaf pine (*Pinus echinata* Mill.), and in general, no clearcutting. The South Alum Creek watershed (SA) is 1460 ha and is the least intensively managed watershed of the four. SA includes an experimental forest and with the exception of a few small experimental cutting treatments, it consists of mature second growth pine-hardwood forest. NA, BC, and SA are all located in Saline County, AR in the Saline River drainage basin. LG is located in Garland County, AR in the Ouachita River drainage basin. Detailed descriptions of these watersheds are found elsewhere in this publication (Tappe and others, in press).

This study was designed to supplement intensive studies of reptile and amphibian communities on the four watersheds (Fox and others, in press; Shipman and others, in press). These studies were designed to compare diversity, richness and abundance of all herpetofaunal species. Because their methods did not adequately sample aquatic turtles, this trapping effort was initiated.

METHODS

Turtles were trapped with baited hoop nets (Lagler 1943, Legler 1960). In 1995, chicken liver and strawberries were used as bait. In 1996, fish and creamed corn were used. Nets consisted of 2.5-cm nylon mesh stretched over 3 hoops of galvanized steel 75 cm in diameter. Nets were approximately 2 m in length. They had a funnel entrance at the downstream end and were closed at the other end. Nets were placed in pools that were deep enough to submerge the bait but shallow enough to allow for breathing space at the top, and were associated with cover (undercut banks, woody debris, rocks) when possible. In 1995 only, ponds were sampled in the LG and NA watersheds. The SA and BC watersheds lacked ponds large enough to allow trapping.

Each trap was kept in the same location for five nights in 1995 and six nights in 1996. Traps were checked daily and bait replaced as needed. All trapping occurred between 11 July and 29 July 1995, and between 15 July and 19 August 1996. Trapped turtles were identified to species, given a unique mark by notching the carapace with a triangular file, measured with tree calipers to the nearest half-centimeter, and released where caught. Exceptions were one razorback

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musk turtle (*Sternotherus carinatus* Gray) that was donated to the Arkansas State University vertebrate museum on request, and one common map turtle (*Graptemys geographica* LeSueur) that appeared to have drowned in the trap.

Habitat data were collected at all 1996 trapping locations. Pool width and length, and maximum depth of each pool were combined into an index of pool size. Substrate compositions of each pool were estimated as percentages in each of four substrate classes: bedrock/boulder, cobble/gravel/sand, silt/mud, or detritus. Boulders were defined as rocks greater than 30 cm diameter. Rocks of smaller size were classified as cobble or gravel. The presence of one or two undercut banks was noted, as was the presence of accessible basking sites (large logs, large rocks).

Relationships between 1996 turtle captures and habitat features were investigated using the SAS PROC CORR procedure (SAS Institute 1993). Pearson correlation coefficients between all habitat variables and number of 1996 turtle captures (all species, excluding recaptures) were calculated.

RESULTS AND DISCUSSION

In 1995, 64 trap-nights were completed in streams and 20 in ponds. In 1996, 128 trap-nights were completed, all in streams. Fifty-five individual turtles representing 6 species were captured a total of 63 times over the 2 years (table 1). NA and LG watersheds had the richest turtle fauna with four species each, and these watersheds also had the largest number of captures, with 19 and 29, respectively. The two watersheds representing the highest level of management intensity therefore produced the most abundance and richness. This relationship is obscured by the fact that these are also the largest watersheds. It is clear that the presence of large ponds in NA and LG added captures, and these ponds are a direct result of management.

The most frequently captured species was the common snapping turtle (*Chelydra serpentina* L.), with 31 individuals captured 39 times in all four watersheds. The presence of ponds in NA and LG contributed to these watersheds having large numbers of common snapping turtle captures. Eleven snapping turtles were captured in 20 trap-nights in ponds (0.55 captures per trap-night). In streams, 28 common snapping turtles were captured in 192 trap-nights (0.15 captures per trap-night). Although it is a habitat generalist (Ernst and others 1994), ponds may be an important habitat feature for this species.

All recaptures were common snapping turtles. This may indicate a strong bond to home range (Ernst and others 1994). In fact, two individuals were captured in the same location in both years. A third individual captured in 1995 moved upstream a short distance and was recaptured in 1996. All other recaptures involved turtles that were released from a trap and then caught again in the same trap within a few days.

Nine razorback musk turtles were captured, all in the Saline County watersheds. This species is near its northern limit in the Ouachitas, and these appear to be the first documented records for this county. No razorback musk turtles were captured in Garland County, although four previous records exist (Personal communication. 1998. Stan Trauth. Arkansas State University, State University, AR 72467). Seven common musk turtles (*Sternotherus odoratus* Latreille) were captured among three watersheds.

All five captures of common map turtles occurred in LG, the only watershed in the Ouachita River drainage basin. Besides geographic isolation, there may be subtle physical differences that make LG a better habitat for this species. Habitat features that may be important to common map turtles include rocky or gravelly substrates (Fuselier and

Table 1—Aquatic turtles captured with hoop-nets in four Ouachita Mountain watersheds by species (Arkansas), 1995 and 1996 combined data

	Watershed				
Species	NA	LG	SA	BC	Total
Common snapping turtle	4 + 3[a]	13 + 8	7	4	28 + 11
Razorback musk turtle	8	0	0	1	9
Common musk turtle	3	1	3	0	7
Common map turtle	0	5	0	0	5
Red-eared slider	0	0 + 2	0	0	0 + 2
Alligator snapping turtle	1	0	0	0	1
Total	16 + 3	19 + 10	10	5	50 + 13

NA = North Alum Creek; LG = Little Glazypeau Creek; SA = South Alum Creek; BC = Bread Creek. Watersheds listed in order of decreasing size.

[a] Data are combined for ponds (20 trap-nights in NA and LG) and streams (192 trap-nights in all watersheds). Where pond captures are present, data presented as (stream captures) + (pond captures).

Edds 1994), in-stream basking sites (Pluto and Bellis 1986), and plentiful mollusk food items (Vogt 1981). Red-eared sliders (*Trachemys scripta elegans* Wied-Neuwied) were also captured only in LG, and these were only in ponds. This species is known to occur in both streams and ponds, but prefers slow-moving or still waters with muddy bottoms (Ernst and others 1994).

One alligator snapping turtle (*Macroclemys temminckii* Harlan) was captured, in the largest pool in NA. This species is protected by the states of Arkansas and Oklahoma, and is thought to be in decline throughout its range. Possible factors include over-harvest and impoundment of rivers (George 1987, Pritchard 1989).

Pearson correlation coefficients relating habitat variables to number of 1996 captures are shown in table 2. Number of 1996 turtle captures was significantly correlated only with the index of pool volume, indicating that larger pools produced more captures. Percent cobble/gravel/sand was significantly negatively correlated with percent rock/bedrock and percent detritus. There are several possible reasons for the relationship between pool size and turtle captures. Larger pools are more persistent throughout the dry summer season. They provide greater cover in the form of deep water than smaller pools. They may provide a greater diversity of habitats such as different water depths. Finally, they may not provide better habitat, but just more of it. If each trap tended to attract all the turtles from the pool in which it was placed, it is obvious that traps in larger pools would attract more turtles.

The lack of correlation between basking sites and turtle captures may be due to the heavy bias towards common snapping turtles, which do not usually bask, and to the use of presence/absence data, rather than number or density of basking sites. There were very few large logs in the stream channels near trapping sites. This is a potential area where turtle habitat could be improved actively in all four watersheds. Besides serving as basking and hiding habitat for turtles (Pluto and Bellis 1986), large logs contribute to pool formation and encourage growth of macro-invertebrates that form the basis of aquatic food webs (Hilderbrand and others 1997). As the stream-side forests mature into old-growth stands, the input of large logs will increase. Removal of timber near streams should be restricted, and managers should consider adding woody debris to the streams actively.

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Table 2—Pearson correlation coefficients (and p values) relating habitat features to each other and to number of aquatic turtles captured at 32 trapping sites, Ouachita Mountains, Arkansas, 1996

	Captures	Volume	Percent cobble	Percent rock	Percent detritus	Percent silt	Bank	Bask
Turtles captured (no.)	1.0 (p = 0.0)	0.47612 (p = 0.0059)	0.10530 (p = 0.5663)	- 0.06058 (p = 0.7419)	- 0.09431 (p = 0.6077)	0.07240 (p = 0.6938)	- 0.20777 (p = 0.2539)	0.00242 (p = 0.9895)
Pool volume index		1.0 (p = 0.0)	0.07619 (p = 0.6785)	- 0.12352 (p = 0.5006)	0.07044 (p = 0.7017)	- 0.06625 (p = 0.7186)	0.10908 (p = 0.5523)	0.13873 (p = 0.4489)
Cobble in substrate (%)			1.0 (p = 0.0)	- 0.77357 (p = 0.0001)	- 0.46312 (p = 0.0076)	0.02425 (p = 0.8952)	0.09338 (p = 0.6112)	- 0.01367 (p = 0.9408)
Rock in substrate (%)				1.0 (p = 0.0)	- 0.17032 (p = 0.3513)	- 0.13039 (p = 0.4769)	- 0.20775 (p = 0.2539)	0.17730 (p = 0.3316)
Detritus in substrate (%)					1.0 (p = 0.0)	- 0.13730 (p = 0.4537)	0.17483 (p = 0.3386)	- 0.24942 (p = 0.1686)
Silt in substrarte (%)						1.0 (p = 0.0)	- 0.22504 (p = 0.2156)	0.07059 (p = 0.7010)
Banks overhanging (no.)							1.0 (p = 0.0)	0.14832 (p = 0.4179)
Presence/absence of bask site								1.0 (p = 0.0)

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Session 7

Phase III Vegetation and Social Science Research

Alan Newman, Moderator

BASELINE OVERSTORY CONDITIONS IN FOUR WATERSHEDS OF VARYING MANAGEMENT INTENSITY IN THE EASTERN OUACHITA MOUNTAINS

James M. Guldin and Thomas Foti[1]

Abstract—Baseline tree data were collected in four watersheds in the eastern Ouachita Mountains during 1996-98. By watershed, average basal area ranged from 71 to 102 square feet per acre, average tree density ranged from 234 to 295 trees per acre, and quadratic mean diameter of trees ranged from 7.41 to 8.22 inches. Variables for which the largest values were associated with the unmanaged South Alum Creek watershed and the smallest values were associated with the most intensively managed Little Glazypeau Creek watershed include basal area for all trees, stem density for all trees, basal area and stem density of the white oak-southern red oak species group and for all hardwoods, and basal area and stem density of shortleaf pine and for all conifers. Conversely, variables for which the largest values were associated with the Little Glazypeau watershed, and the lowest values were associated with the South Alum Creek watershed, were loblolly pine basal area and stem density. The data suggest that it may be possible to relate management intensity to descriptive mensurational variables at the landscape scale in the eastern Ouachita Mountains.

INTRODUCTION

The phase III landscape study in the Ouachita Mountains Ecosystem Management Research Project is designed to describe changes in baseline conditions among four watersheds under different intensities of management. The four watersheds under study and their attributes are listed in table 1. An overall description of the study design is presented in Guldin (in press).

Experimental replication of watersheds of substantial size is impractical; it is impossible to achieve uniformity of topographic, edaphic, and physiographic conditions in such watersheds. Some other approach to the development of experimental error is needed if it is necessary to apply parametric statistical tests in this situation. We tackled this question with a study design that used time, rather than space, as an experimental replicate. Thus, 4 years of measurements were taken in these areas, and comparisons among watersheds included variation by year to determine whether treatment or year was the more significant variable.

The objectives of the research reported here are to quantify existing vegetation conditions of pine, pine-hardwood, and hardwood stands in the four watersheds; to compare differences in stand structure by watershed and year; and to describe the range of variation in forest conditions among watersheds by year and treatment.

METHODS

Sample Grid

The baseline overstory vegetation data was collected by systematic sampling over a 4-year period (1995-98). Sampling was conducted using plots on a rectangular grid, with offsets in the grid by year (fig. 1). The grid used as a basis for sampling in 1995 and 1996 consisted of north-south transects 400 m apart. To facilitate access, each transect was flagged and numbered wherever it crossed as road. Additional north-south transects, offset 200 m from those employed in 1995 and 1996, were used in 1997 and 1998.

Sample points were located at 200-m intervals along the first set of transects in the first year. In the second year, the sample points were offset 100 m from the first set of sample points along the same transects. In the third year, sample points were located at 200-m intervals along the second set of transects. In the fourth year, sample points were offset 100 m from the third set of sample points along each of the second set of transects.

Table 1—Attributes of the four watersheds under study

Watershed	Ownership FS	Ownership Industry	Management intensity	Area
	- - *percent* - -			*ha*
Alum Creek	100	0	None	1460
Bread Creek	95	5	Low	1255
North Alum Creek	50	50	Moderate	3080
Little Glazypeau Creek	5	95	High	2450

FS = Forest Service.

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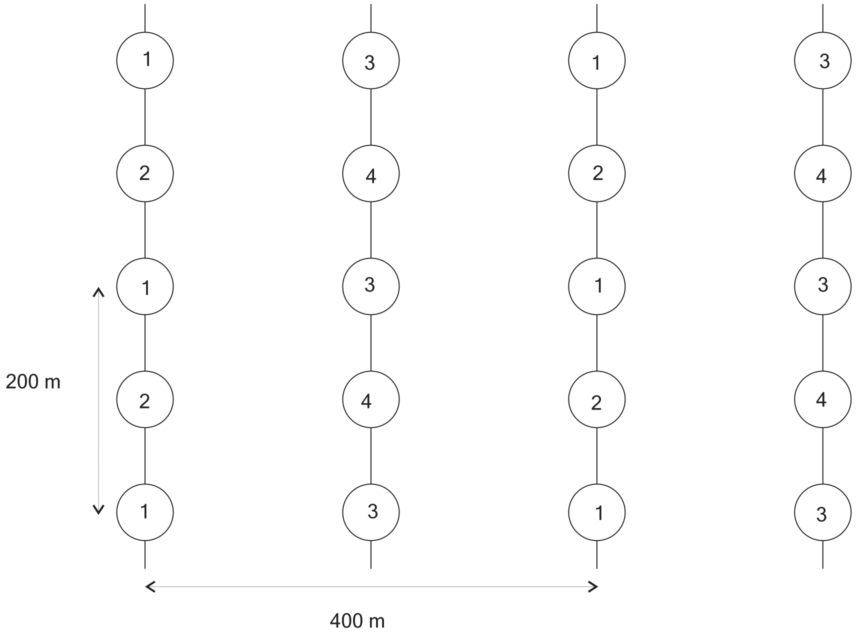

Figure 1—Layout of transects from 1995-98. Plots labeled 1 were sampled in 1995, 2 in 1996, 3 in 1997, and 4 in 1998.

These grid points became the plot centers for the plot-based measurements of vegetation. Approximately 500 plots were sampled annually.

Plot Layout

Two different sampling methods were used. In 1995, stands were sampled using variable radius plots, with a basal area factor (BAF) 10 square-feet-per-acre prism (fig. 2a). All trees 3.0 inches in diameter at breast height (d.b.h.) and larger that were sampled were recorded by species and 2-inch diameter classes. Examination of the data after the first year of sampling suggested that this sampling might not yield data that adequately characterize the vegetation in the watersheds. Because of this concern, sampling in 1996-98 was done using a nested plot design (fig. 2b). Circular 0.1-acre fixed-radius plots were used to sample all trees having d.b.h. from 3.6 to 9.5 inches inclusive, and a BAF 5 square-feet-per-acre prism was used to sample all trees with d.b.h. 9.6 inches and larger. Species and d.b.h. to the nearest 0.1 inch were recorded for all sampled trees.

Statistical Analysis

Two statistical questions arise. First, because two sampling schemes were employed, we have to ask whether the 1995 data can be combined with 1996-98 data. Second, we have to ask whether the replication approach employed was sensitive enough to yield useful results.

Statistical tests suggest that the total basal area variable is not normally distributed under either sample design (the Shapiro-Wilks W statistic was significant at the 0.0001 level for both). For this reason, comparisons of data obtained by different sample methods were conducted using nonparametric analysis of variance, using the Wilcoxon statistic to test significance of mean comparisons. Descriptive baseline information (basal area per acre, trees per acre, and quadratic mean diameter) will be useful to the research team and others.

RESULTS

Comparison of Sampling Methods

The 1995 sample gives a basal area distribution sensitive only to the nearest 10 square feet of basal area (fig. 3a). This is an artifact of the expansion of a prism sample to a per-acre basis. Each tree sampled counts as 10 square feet of basal area per acre in trees of that size class. Therefore, the prism sample taken in 1995 produces a distribution of basal area that is obviously discontinuous. Conversely, the 1996-98 sampling scheme, which combined prism sampling and the use of fixed-radius plots, produces a more continuous distribution (fig. 3b). The shape of the distribution is similar to that of the 1995 sample, but the nested design results in a more uniform distribution of observations across the range of basal areas.

Discontinuities in data, such as that found in the 1995 sample, can violate underlying assumptions in parametrical statistical tests. One such test suggests that total basal area variable is not normally distributed under either sample design (the Shapiro-Wilks W statistic was significant at the 0.0001 level for both). For this reason, comparisons between variable means for data obtained by different sample methods were conducted using nonparametric analysis of variance. The Wilcoxon statistic was used to test significance of mean comparisons. Using this test, significant differences in variable means by sample design were detected for virtually all summary variables of basal area and stem density (table 2). Four of the six species groups show significantly higher basal area in the 1996-98 sample design than the 1995 sample design.

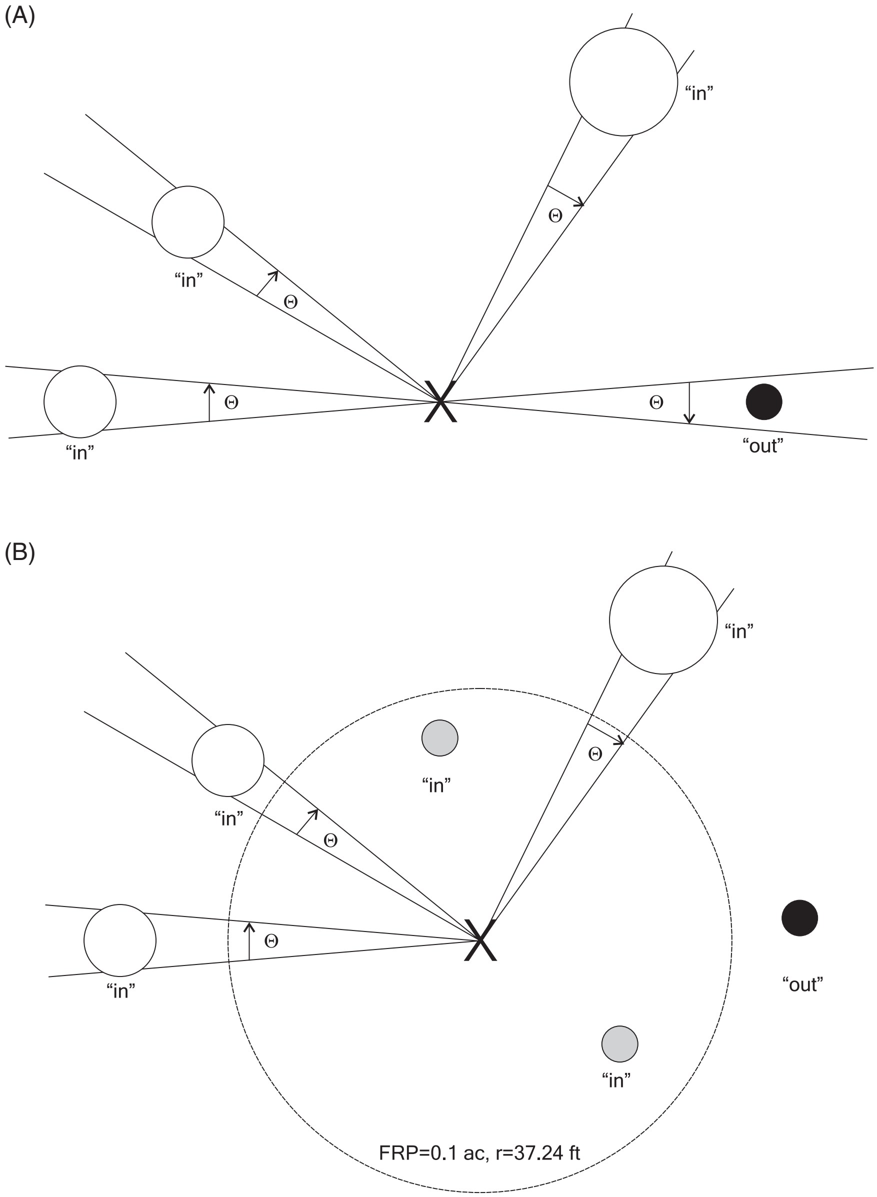

Figure 2—Overstory vegetation sampling methods used in (A) 1995, and (B) 1996-98. (A) 1995 sampling. Sampling is conducted using a variable radius prism where Θ corresponds to a basal area factor (BAF) of 10 square feet per acre, which marginally samples a tree with diameter at breast height (d.b.h.) 12.0 inches at 33 feet; and (B) 1996-98 sampling. Sampling is based on use of a nested fixed radius plot (FRP) of 0.1-acre to sample midstory trees having d.b.h. 3.6 and 9.5 inches inclusive, combined with use of a prsim where Θ corresponds to a BAF of 5 square feet per acre to sample trees with d.b.h. greater than or equal to 9.6 inches. A BAF 5 prism marginally samples a 12.0-inch tree at 46.67 feet.

The largest difference is in the sample for loblolly pine, which is found exclusively in plantations in these watersheds. Loblolly pine accounts for only 10 percent of the basal area in the 1995 sample, but the difference in the loblolly pine basal area between the two samples is 68 percent of the total difference between the samples. These loblolly pine plantations are often very dense and difficult to see through, which may contribute to inaccuracy in the prism sample.

Of the three broad species groups (all conifers, all hardwoods, and all species), the all conifers and all species groups show significantly higher basal area in the 1996-98 sample than in the 1995 sample (table 2). Four of the six species groups show significantly higher stem density in the 1996-98 sample than in the 1995 sample, as do three broad summary variables.

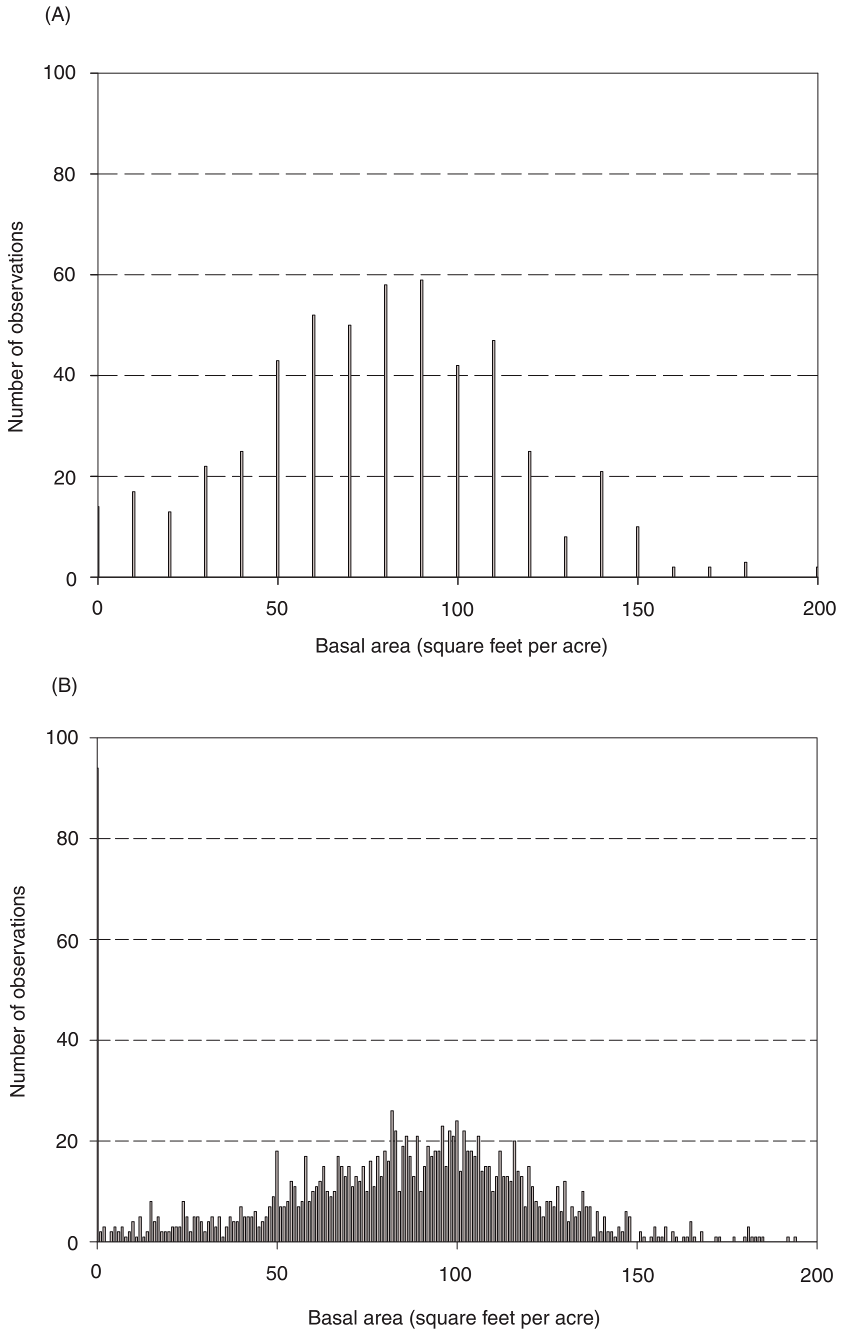

Figure 3—Number of observations by 1-foot basal area increments, 1995 sample design versus 1996-98 sample design. (A) 1995 sample design, and (B) 1996-98 sample design.

To determine the effect of sample design on variable means, we randomly selected and remeasured eighty-one 1995 plots in 1997 according to the 1996-98 sample design. In all instances, broad summary variables have higher means and lower coefficients of variation in the 1996-98 sample design than in the 1995 sample design (table 3). Results for two important species or species groups are consistent with these findings—shortleaf pine (fig. 4a) and the white oak-southern red oak group (fig. 4b).

Based on these results, it is difficult to justify pooling the 1995 data with the 1996-98 data for future analyses. For the balance of this paper, only the 1996-98 data will be used as a basis for discussing the conditions in the watersheds.

Overall Stand Conditions

The mean basal area (all species combined) across all watersheds over 1996-98 was 86.4 square feet per acre. Conifer basal area across all watersheds averaged 49.5 square feet per acre, of which 35.8 square feet per acre was shortleaf pine and 13.7 square feet per acre was loblolly, with

Table 2—Differences in mean species group summary variables, by sample design, across all watersheds[a]

	Sample design		
	1995	1996–98	Prob > |Z|
	- - - ft^2 per acre- - -		
A. Basal area			
Shortleaf pine	34.32	35.75	NS
Loblolly pine	8.28	13.74	0.001
Other conifers	0.00	0.05	0.005
White and southern red oak	16.32	17.53	0.0001
Other oaks	10.12	9.31	0.01
Other hardwoods	9.40	10.02	0.0001
All conifers	42.60	49.54	0.0001
All hardwood	35.83	36.86	NS
All species	78.43	86.39	0.0001
	Trees per acre		
B. Stem density			
Shortleaf pine	94.97	95.26	0.05
Loblolly pine	40.52	46.16	0.01
Other conifers	0.00	0.21	0.01
White and southern red oak	60.22	58.97	0.0001
Other oaks	31.03	30.01	0.0001
Other hardwoods	41.04	47.84	0.0001
All conifers	135.48	141.64	0.0001
All hardwoods	132.28	136.82	0.001
All species	267.77	278.46	0.0001

[a] Differences between means are shown by Prob > |Z|, the nonparametric Wilxocon 2-sample test statistic with normal approximation; the value shown is the probability level at which the null hypothesis of no difference between means is rejected.

Table 3—Mean and coefficient of variation for summary basal area and stem density variables for subset of 1995 sample design plots that were remeasured in 1997 using the 1996–98 sample design

	Sample design			
	1995		1996–98	
	Mean	C.V.	Mean	C.V.
	ft^2/ac	*percent*	*ft^2/ac*	*percent*
A. Basal area				
All conifers	48.89	69.1	58.29	61.7
All hardwoods	31.72	80.8	40.32	60.9
All species	80.62	43.3	98.61	33.4
	trees/ac	*percent*	*trees/ac*	*percent*
B. Stem density				
All conifers	139.2	109.4	138.2	82.6
All hardwoods	122.4	83.2	161.4	50.8
All species	261.5	63.5	299.7	37.0

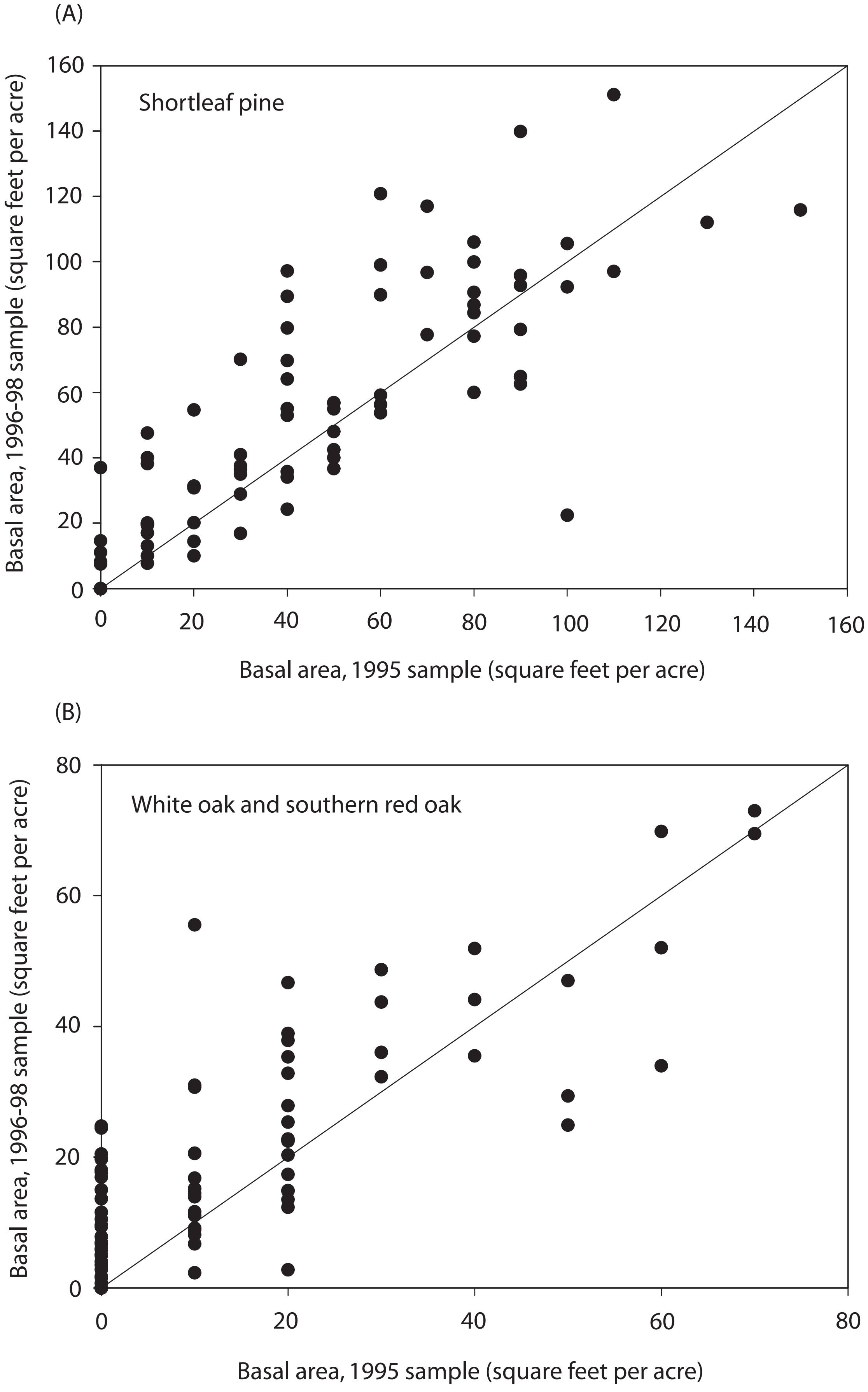

Figure 4—Basal area by plot, for the 81-plot subset of 1995 sample design plots that were remeasured in 1997 using the 1996-98 sample design. (A) shortleaf pine, and (B) white oak and southern red oak.

the small balance in eastern redcedar. The mean hardwood basal area across all watersheds was 36.9 square feet per acre, of which nearly half (17.5 square feet per acre) was in white and southern red oak, and the balance more or less equally split between other oaks (9.3 square feet per acre) and other hardwoods (10.0 square feet per acre).

Mean stem density (all species combined) across all watersheds over 1996-98 was 278.5 trees per acre, nearly equally split between conifers (141.6 trees per acre) and hardwoods (136.8 trees per acre). Shortleaf pine was most numerous, with 95.3 trees per acre, followed by white and southern red oaks (59.0 trees per acre), other hardwoods (47.8 trees per acre), loblolly pine (46.2 trees per acre), other oaks (30.0 trees per acre), and eastern redcedar (0.2 trees per acre).

South Alum Creek

The unmanaged South Alum Creek watershed had, on average, 295 trees per acre (table 4). Of these, 174 trees per acre were hardwoods and 121 trees per acre were conifers. Almost all of the conifers that were present were shortleaf pine.

Basal area (all species combined) was higher (102.5 square feet per acre) than it was in the other watersheds studied (table 4). Of this, 55.4 square feet per acre was in conifers and 47.1 square feet per acre was in hardwoods. Shortleaf pine made up more than 99.9 percent of the conifer basal area in the South Alum Creek watershed. The remaining 0.1 percent was loblolly pine, recorded in a handful of research plots that were planted to loblolly in the 1970s and 1980s. No eastern redcedar was sampled during the 3 years of data collection.

Over half (56.6 percent) of the hardwood basal area in the South Alum Creek watershed was in white and southern red oaks. The remaining basal area consisted almost equally of other oaks (primarily black and post oaks) and hardwoods other than oak.

Quadratic mean diameter (qdbh) is the diameter calculated from the mean basal area per tree. In the South Alum Creek watershed, the qdbh for all species combined during 1996-98 was 8.23 inches. Conifers were, on average, larger than hardwoods in this watershed; the qdbh for conifers was 10.59 inches, whereas that for hardwoods was 7.06 inches.

Bread Creek

The Bread Creek watershed, characterized by general Forest Service management, supported an average of 292.0 trees per acre (table 4), exactly divided between hardwoods and conifers. Of the total, over 70 percent of stems were shortleaf pine, white oak, or southern red oak. More than 97 percent of the conifers present were shortleaf pine, and more than 70 percent of the hardwoods present were oaks.

Basal area averaged 86.2 square feet per acre (table 4), of which 49.7 square feet per acre was in conifers and 36.5 square feet per acre was in hardwood. Shortleaf pine accounted for 97.1 percent of the conifer basal area. Loblolly pine, in plantations, made up 2.8 percent of conifer basal area. White oak and southern red oak constituted 52.4 percent of hardwood basal area, with an additional 26.6 percent in other oak species and the balance in other hardwoods.

In the Bread Creek watershed, qdbh for all trees sampled was 7.4 inches, which was smaller than qdbh in the other watersheds. Conifer qdbh was 8.5 inches. The qdbh of

Table 4—Mean basal area and stem density by species or species group and watershed, 1996–98

	South Alum Creek	Bread Creek	North Alum Creek	Little Glazypeau Creek
A. Basal area				
Shortleaf pine	55.43	48.26	35.54	15.92
Loblolly pine	0.02	1.39	14.26	29.01
Other conifers	0.00	0.03	0.09	0.01
White oak-southern red oak	26.64	19.11	18.54	8.88
Other oaks	11.28	9.72	9.82	6.90
Other hardwoods	9.14	7.65	10.57	11.02
Total	102.51	86.15	88.82	71.75
B. Stem density				
Shortleaf pine	120.61	142.18	107.32	30.90
Loblolly pine	0.20	3.52	42.46	107.36
Other conifers	0.00	0.28	0.37	0.05
White oak-southern red oak	94.33	67.31	61.82	25.98
Other oaks	39.05	36.83	30.50	19.28
Other hardwoods	41.05	41.85	50.92	50.47
Total	295.24	291.96	293.39	234.04

shortleaf pine in this watershed, 8.5 inches, was a full inch less than qdbh for shortleaf in any of the other watersheds. The mean qdbh of hardwoods in the Bread Creek watershed, 6.6 inches, was less than qdbh of hardwoods in any other watershed; and qdbh of white and southern red oaks, 7.4 inches, was 0.1 inch less than qdbh for these species in any other watershed.

North Alum Creek

Industry owns half of the North Alum Creek watershed, and the Forest Service owns the other half. Tree density averaged 293.4 trees per acre during 1996-98 (table 4). Density of conifers was 150.1 trees per acre, and density of hardwoods was 143.2 trees per acre. The industry presence accounts for the importance of loblolly pine, which constituted 28.3 percent of conifer stem density in the watershed.

Basal area in the North Alum Creek watershed averaged 88.8 square feet per acre (table 4). Of this, 49.9 square feet per acre was in conifers and 38.9 square feet per acre in hardwoods. Shortleaf pine accounted for 71.2 percent, and loblolly for 28.6 percent, of conifer basal area. Oaks constituted 72.8 percent hardwood of basal area, with white and southern red oak accounting for nearly two-thirds of the oak basal area.

The qdbh of trees in the North Alum Creek watershed was 7.8 inches. Conifer qdbh was 9.5 inches and hardwood qdbh was 7.1 inches. The quadratic mean diameters of shortleaf and loblolly pine were 9.6 and 8.4 inches, respectively. The quadratic mean diameters of the white oak-southern red oak group and the other oak group were 8.0 and 8.3 inches, respectively.

Little Glazypeau Creek

The Little Glazypeau watershed, managed largely by forest industry, had 234.0 trees per acre (table 4), about 20 percent fewer stems per acre than were present in the other three watersheds. The reduction was in hardwood density, where declined by more than one-third as density of white and southern red oaks declined by nearly two-thirds.

Basal area in the Little Glazypeau Creek watershed averaged 71.7 square feet per acre (table 4); of this, 44.9 square feet per acre was in the conifer component and 26.8 square feet per acre was in the hardwood component. Conifer basal area in this watershed was about 12 percent lower than that in the other three watersheds. Loblolly pine constituted two-thirds of conifer basal area in this watershed (compared to one-sixth of average conifer basal area across the other three watersheds). The proportion of hardwood basal area to total basal area was 33 percent lower than in the other watersheds, and basal area in white and southern red oaks was also 33 percent lower in Little Glazypeau than in the other watersheds.

The qdbh of trees in the Little Glazypeau watershed was 8.0 inches. As in the other watersheds, conifer qdbh (9.9 inches) was greater than hardwood qdbh (6.9 inches).

DISCUSSION

We believe that differences between the sample designs employed in 1995 and 1996-98 are significant, and we conclude that the two designs should not be combined. All of the 1995 data were collected by sampling with a BAF 10 square-feet-per-acre prism, and are discontinuous. This discontinuity has adverse implications for conducting statistical analysis that would not apply to the 1996-98 data. More importantly, mean values of stem density and basal area for important species and species groups from the 1995 sample are typically lower than corresponding values for the 1996-98 data, a difference greater than that likely to be explained by normal forest growth. For both of these reasons, summary statistics reported here were prepared only on the basis of the nested sample design used in 1996-98.

By watershed, average basal area ranged from 71 to 102 square feet per acre, with the lowest basal area in the most intensively managed Glazypeau watershed and the highest in the unmanaged South Alum Creek watershed. Average tree density varied from 234 to 295 trees per acre, again with the lowest tree density in the Little Glazypeau Creek watershed and the highest in the South Alum Creek watershed. Quadratic mean diameter varied from 7.41 to 8.22 inches. Quadratic mean diameter was least in the Bread Creek watershed, and largest in the South Alum Creek watershed.

Values of other variables discussed here follow a similar pattern of association. The variables for which the largest values are associated with the unmanaged South Alum Creek watershed and the smallest values are associated with the most intensively managed Little Glazypeau Creek watershed include the following

- basal area of shortleaf pine
- basal area of all conifers
- basal area of the white oak-southern red oak group
- basal area of the other oak group
- basal area of all hardwoods
- basal area of all trees
- stem density of the white oak-southern red oak group
- stem density of the other oak group
- stem density of all hardwoods
- stem density of all trees.

Conversely, the variables for which the smallest values are associated with the unmanaged South Alum Creek watershed the largest values are associated with the Little Glazypeau Creek watershed are

- basal area of loblolly pine
- stem density of loblolly pine
- quadratic mean diameter of loblolly pine.

These associations are not unexpected, and reflect the common silvicultural practices one would conduct in managed landscapes. For example, thinning in the conifer component improves the condition of retained conifers by cutting the poorer and less vigorous trees, and reduces density-dependent mortality. Timber stand improvement cutting in the shortleaf pine-hardwood forest type removes hardwoods to release pines, and is often done in conjunction with thinning of pines. The result of these management actions is to reduce both stem density and basal area in the conifer and hardwood component relative to unmanaged stands. Reproduction cutting in the managed watersheds would result in removal of the larger conifers, which would reduce quadratic mean diameter in the managed watersheds. Finally, the most intensive silvicultural practice is to completely remove the shortleaf pine component and replant with loblolly pine, which grows more rapidly than shortleaf pine but which is not naturally found in these watersheds.

In conclusion, these data give some interesting clues about the effects of varying intensities of management on descriptive mensurational statistics used to characterize average stand conditions. More comprehensive analysis of the data is needed to better indicate the robustness of these trends, and to show whether the trends are statistically significant. However, this subjective analysis suggests that it may be possible to quantify landscape attributes associated with increasing management intensity for the woody plant component in the Ouachita Mountains.

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ORDINATION OF WOODY VEGETATION IN A OUACHITA NATIONAL FOREST WATERSHED

Denise Marion and George Malanson[1]

Abstract—Species response to competition and other environmental gradients has important implications for forest ecosystem managers who desire to both maintain diversity and provide a sustained flow of forest goods and services. Woody species on a 140-acre watershed in the Ouachita National Forest are ordinated with detrended correspondence analysis (DCA) and canonical correspondence analysis (CCA) to identify the important influences on species distribution at this scale. Species composition is found to respond primarily to a moisture gradient, and secondarily to competition with shortleaf pine (*Pinus echinata* Mill.). Centrifugal organization of species along gradients is suggested by the relative locations of species in the ordination diagrams.

INTRODUCTION

The objective of ecosystem management is to sustain the structure, function, and diversity of the ecosystem while still providing goods and services to the public. Meeting these goals requires a fuller understanding how environmental factors such as disturbance, soil moisture, and interspecific competition affect the distribution of plant species across the landscape. This study uses multivariate ordination to examine the influence of competition on the pattern of distribution of woody species. The resulting pattern is compared with the centrifugal model of plant community organization proposed by Keddy and MacLellan (1990) (fig. 1).

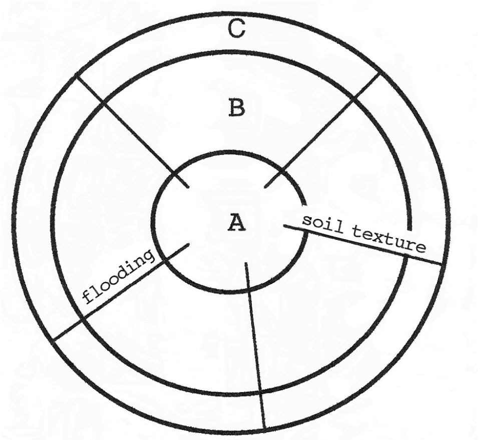

Figure 1—Centrifugal model of plant community organization. Environmental gradients represented by lines radiating out from core; e.g. flooding and soil texture, (A) mesic fertile sites, gap dynamics, one or two dominant species; (B) competitive hierarchy, shade tolerance increases toward core; (C) extreme site conditions, large-scale disturbance, greatest beta diversity (adapted from Keddy and MacLellan 1990).

Keddy and MacLellan's model describes the pattern of species distribution along biomass gradients[2] which radiate out from a central core which is characterized by fertile, mesic, and low disturbance environments. The underlying mechanism in this model is competition for light; consequently the largest species dominate the most productive habitat. Although all species could thrive in the core habitat, most species are prevented from doing so by virtue of being shaded by the large dominant. Species distribution within the core is primarily a result of gap dynamics. At their core ends the biomass gradients have similar species composition:

At the less fertile and/or high disturbance end of the gradients species compositions vary as the environmental stresses change. Less successful competitors for light in the optimal habitat achieve greater importance in more stressful environments where adaptations to specific conditions give them a competitive advantage. A greater beta diversity, or turnover of species, is thus predicted in the extreme environments represented at the periphery of the model.

DOMINANCE OF SHORTLEAF PINE

Shortleaf pine (*Pinus echinata* Mill.) density and volume is highest in the Ouachita Mountains, where it has regenerated naturally from seedlings on cut-over, burned, or abandoned land to dominate natural stands. It dominates the Ouachita pine-oak forest where the colder and drier conditions are not suitable for loblolly, and where the moisture, nutrient, and disturbance regimes discourage the growth of hardwood species (Guldin 1986).

Shortleaf pine is an example of a species which has its greatest abundance outside of its optimal habitat. It is most productive in well drained, sandy loam soils with a low pH and calcium content, on low gradient north aspects, at lower elevation and latitude. In these sites, however, it is outcompeted by faster growing pines or overtaken by longer-lived

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[2] Organization is based on biomass, with the greatest biomass in the optimal habitat. Because the model was originally developed for herbaceous vegetation, biomass was used as a surrogate measure of productivity. This interpretation of biomass does not transfer to woody species, however, where biomass is the accumulation of productivity of years past. Therefore, no conclusions regarding productivity can be inferred from this study.

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hardwoods. Its ecological importance increases in drier, less fertile upland sites where its large root system, lower demand for nutrients, and tolerance of fire and cold give it a competitive advantage. The success of the shortleaf pine in the Ouachitas can be attributed to the location of the uplands relative to the neighboring loblolly-oak forest to the south, and the oak-hickory forest to the north (Burns and Honkala 1990, Guldin 1986).

STUDY SITE

The study site is the north-facing, 56.7 ha (140 acres) watershed located in the shortleaf pine and oak forest of the Ouachita National Forest, northwest of Hot Springs, AR. The watershed was surveyed and sampled by foresters in 1994 as part of a fluvial geomorphology research project (Personal communication. Daniel A. Marion. 1996. Research Hydrologist, USDA Forest Service, Southern Research Station, 1000 Front St., Oxford, MS 38655). Average annual precipitation is 51 inches. Elevation ranges from 160 to 396 m (560 to 1,300 feet), with slope gradients up to 30 percent.

Three soil units occur in the watershed, classified according to the USDA Forest Service, Southern Region Soil Resource Guide and the Natural Resources Conservation Service (NRCS) (Luckow and Lee 1995). The majority of the watershed falls in one soil map unit which is identified as a Pirum-Clebit-Carnasaw complex. These are characteristic of moderately steep mountain side slopes. Soils at the upper elevations on the western divide are moderately deep clayey and loamy soils characteristic of mountain tops. The major component of this unit is a Carnasaw soil. The third soil unit occurs on the highest elevations in the watershed, which form the southeastern divide. It is a Clebit-Carnasaw-Pirum complex characteristic of upper side slopes and mountain tops. About 3 inches of very stony, fine sandy loam overlay a gravelly loam subsoil layer.

Three vegetation patches represent the pattern of disturbance in the watershed: approximately 75 acres are shortleaf pine-hardwood forest, 56 acres are plantation, and 8 acres are riparian. The forest to the west of the creek and in the southern end of the watershed is roughly 70 years old. It has been relatively undisturbed since logging in the early part of this century. Flooding in the riparian zone establishes a 1- to 5-year disturbance regime adjacent to the channel, and a 30- to 50-year regime near the break in the slope. The eastern side of the watershed is a plantation < 10 years free of disturbance. A seed-tree harvest was completed sometime between 1979 and 1982. When the seed trees were removed in 1986 natural reseeding was judged unsuccessful. In 1988 the plantation site was prepared for planting with herbicide suppression of hardwoods and mechanized exposure of bare soil. In 1989 the area was planted uniformly with 1 year-old shortleaf pine seedlings, grown from seeds obtained in the Forest Service's seed orchard in the Ouachita National Forest.

METHODS

All stems >1 m in height are identified, tallied, and measured. Stems < 4 inches in diameter are grouped into one size-class. Sample sites are located by stratified random sampling: 50 percent of the plots are located in the forested area; 30 percent of the plots are located in the clear-cut area; 20 percent of the plots are in the riparian zone. The riparian zone is defined as the land adjacent to the stream channel between the slope breaks; a distance of 5 to 30 feet on either side.

Environmental variables used in the analysis are elevation, percent slope, aspect, height above stream, Natural Resources Conservation Service (NRCS) soil factor T, litter cover, and shortleaf pine importance. Elevation, slope, height above stream, and aspect are taken from the U.S. Geological Survey 7.5 minute Nimrod SE topographic map, revised in 1985. An insolation value for aspect was read from a table calculated by Beers (1966). NRCS factor T is an estimate of the maximum average annual rate of soil erosion by wind or water that can occur without affecting crop productivity over a sustained period. Litter cover is the percentage of ground with no bare mineral exposed, estimated visually. Importance values are computed as each species' percent basal area plus its percent of individuals. In order to evaluate the effect of competition with shortleaf pine its importance value is removed from the dependent (species) variables and used as an environmental variable.

ORDINATION

Ordination is the ordering of species characteristics, such as abundance or productivity, along an environmental gradient, or a transect thought to represent a change in environmental conditions. The goal of ordination is to identify important environmental influences on plant community composition. An ordination diagram displays compositionally similar sites near each other; dissimilar sites are located farther apart. Multivariate ordination represents species or site data in two or more dimensions where axes represent complex gradients. The first axis explains the greatest proportion of variance in the data, the second axis explains the next greatest proportion, and so forth for additional axes.

Indirect ordination techniques, such as detrended correspondence analysis (DCA) order only species and sample plots without direct information regarding habitat conditions. Environmental variables are then correlated with the ordination in a second step. In direct ordination, such as canonical correspondence analysis (CCA), axes represent a linear combination of environmental variables that optimize the separation of the species response curves. In a CCA graph, environmental gradients are indicated by arrows pointing in the direction of the greatest rate of change. Arrows pointing in a similar direction are highly correlated; arrows pointing in opposite directions indicate a high negative correlation (ter Braak and Prentice 1988). The length of the arrow is proportional to the strength of the correlation with the ordination axes. The length is, therefore, an indicator of the strength of the relationship of the environmental variable to the pattern of community variation shown in the graph (ter Braak 1987).

In both DCA and CCA ordination graphs, points that represent sample sites are located at the center of gravity of the species that occur there. Species points at the edges of a DCA ordination are often rare species occupying extreme environmental conditions (ter Braak and Prentice 1988).

Data are ordinated with PC-ORD (McCune and Mefford 1995). DCA ordination is run with all species. CCA ordinations are run with all species, without shortleaf pine, and with shortleaf pine as an environmental (independent) variable. Shortleaf pine is removed from the species variables to allow evaluation of the effect of shortleaf pine competition as an environmental variable. Rare species are not downweighted. DCA and CCA ordination scores are correlated with environmental variables in PC-ORD with both Pearson's and Kendall's correlation coefficients.

RESULTS

Importance Values

Table 1 lists the woody species identified in the watershed. Shortleaf pine dominates the watershed as a whole at with an importance value of 48 percent. Only five sample plots in the watershed are dominated by species other than shortleaf pine: two are plantation sites where snowbell and red maple (*Acer rubrum* L.) reach high importance; the others are riparian sites where ironwood (*Ostrya virginiana* Scop.), and sometimes sweetgum (*Liquidambar styraciflua* L.) and/or blackgum (*Nyssa sylvatica* Marsh.) are more important than pine. Shortleaf pine dominance is greatest in the forest patch where its importance reaches 48 percent, and its closest competitor, white oak, achieves only 12 percent (fig. 2). In the plantation patch, the importance of shortleaf pine is 28 percent, whereas both snowbell and hickory (*Carya* spp.) achieve 15 percent importance. In the riparian patch

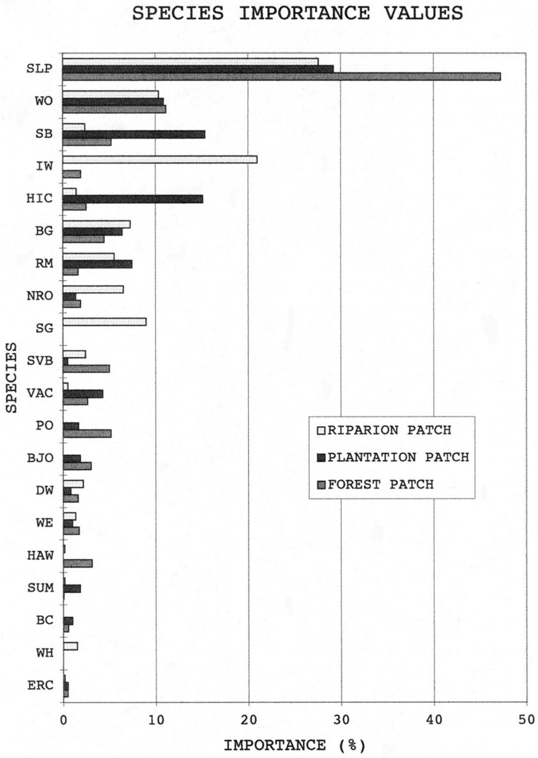

Figure 2—Species importance values by patch type. Species with importance values <1 percent are omitted.

Table 1—Species and code names in Toots Creek watershed survey

Code	Scientific name	Common name
BC	*Prunus serotina* Ehrh.	Black cherry
BG	*Nyssa sylvatica* Marsh	Blackgum
BH	*Viburnum rufidulum* Raf.	Rusty blackhaw
BJO	*Quercus marilandica* Muenchh.	Blackjack oak
BO	*Quercus velutina* Lam.	Black oak
ERC	*Juniperus virginiana* Alt	Eastern red cedar
DW	*Cornus florida* L.	Flowering dogwood
FM	*Acer barbatum* Michx.	Florida maple
HAW	*Crataegus* spp.	Hawthorn
HBM	*Carpinus caroliniana* Walt.	Hornbeam, blue beech
HIC	*Carya* spp.	Hickory
IW	*Ostrya virginiana* Scop.	Ironwood, eastern hophornbeam
NRO	*Quercus rubra* Michx. f.	Northern red oak
PL	*Prunus* spp.	Plum
PO	*Quercus stellata* Wangenh.	Post oak
RM	*Acer rubrum* L.	Red maple
SB	*Styrax grandifolius* Ait.	Bigleaf snowbell
SG	*Liquidambar styraciflua* L.	Sweetgum
SLP	*Pinus echinata* Mill.	Shortleaf pine
SUM	*Rhus* spp.	Sumac smooth, shining
SVB	*Amelanchier* spp.	Service berry
VAC	*Vaccinium* spp.	Farkleberry, huckleberry
WA	*Fraxinus americana* L.	White ash
WE	*Ulmus alata* Michx.	Winged elm
WH	*Hamamelis virginiana* L.	Witch-hazel
WIO	*Quercus phellos* L.	Willow oak
WO	*Quercus alba* L.	White oak
WTO	*Quercus nigra* L.	Water oak

shortleaf pine's importance value, 28 percent, is only slightly greater than that of ironwood at 22 percent. White oak also reaches 10 percent importance in the riparian patch.

CCA

Figure 3 shows the results of CCA ordination in three different sets: shortleaf pine as a species variable but not an environmental variable; shortleaf pine omitted from both the species variables and environmental variables; and shortleaf pine employed as an environmental variable rather than a species variable. Table 2 lists correlation coefficients between the axes and the environmental variables.

In the all-species ordination (fig. 3A) forest sites are in the center, and riparian and plantation sites are distributed on either end of the first axis. The first CCA axis is most strongly correlated with height above the stream (r = .786), which represents a moisture gradient. The second axis is correlated most strongly with soils (r = -.782). The pattern of distribution of the sites in ordination space is altered radically when pine is removed from the environmental variables (fig. 3B). All sites except three, four, and six are tightly clustered in the middle of axis two, stretched in a slightly negative trend along axis one. Species composition appears to be influenced primarily by moisture and soil gradients. Patch types remain in the same relative locations as in the all-species analysis.

When shortleaf pine is used in the analysis as an environmental variable (fig. 3C) the central cluster is scattered vertically by the second axis, which is now most strongly correlated with importance value of shortleaf pine (r = -.652). The spread of the plantation group is nearly parallel to the shortleaf pine vector, indicating a strong association with this environmental variable. The limited change in the riparian group indicates that shortleaf pine has less effect on species composition in the riparian sites than in the forest and plantation sites.

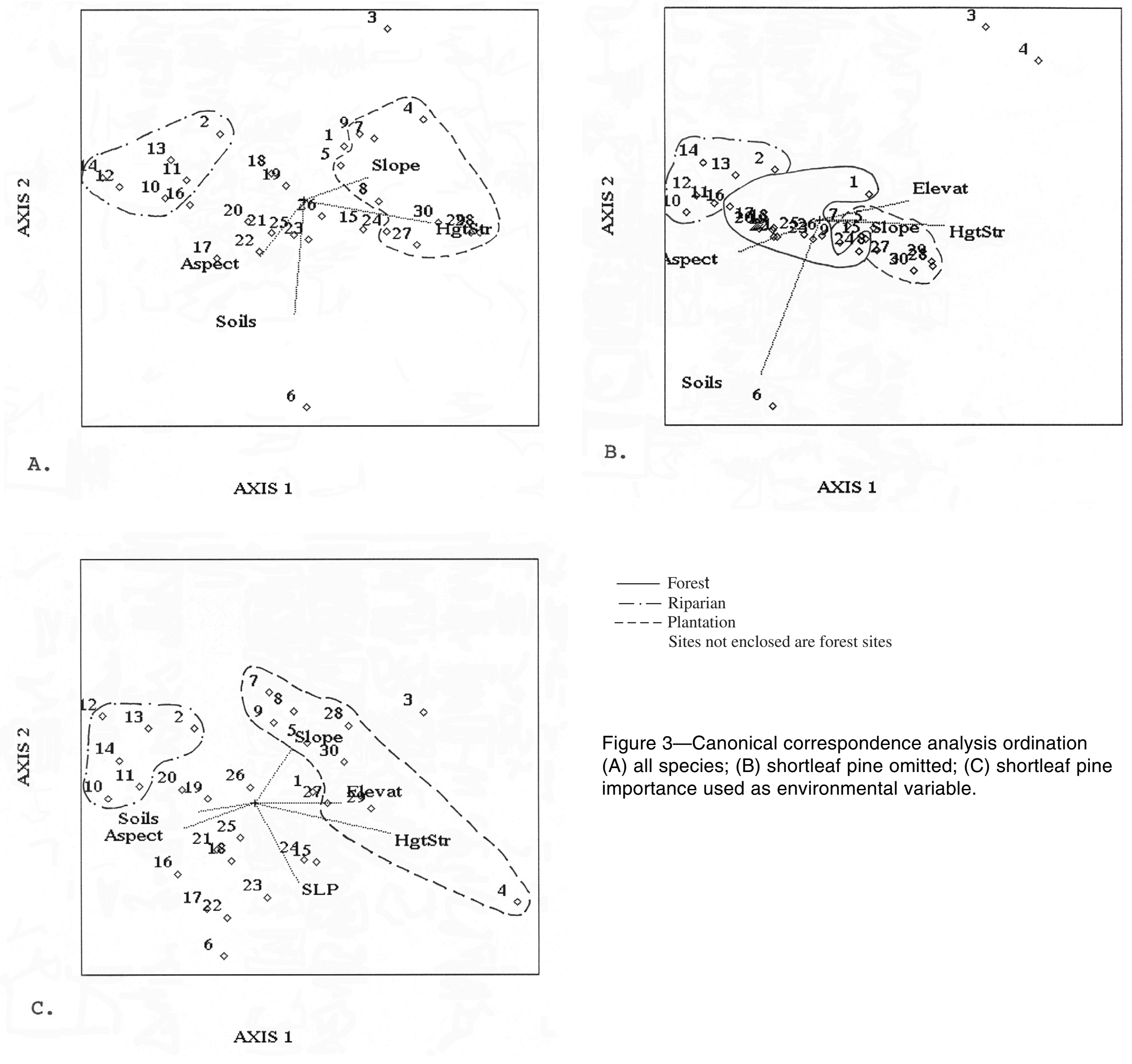

Figure 3—Canonical correspondence analysis ordination (A) all species; (B) shortleaf pine omitted; (C) shortleaf pine importance used as environmental variable.

Table 2—Correlation of environmental variables with CCA axes (Pearson's r)

	All species			No pine			Pine as environmental variable		
	Axis1	Axis2	Axis3	Axis1	Axis2	Axis3	Axis1	Axis2	Axis3
Height above stream	.786	-.344	.437	.777	-.166	.558	.809	-.402	.096
Soils	-.214	-.782	-.486	-.534	-.904	-.173	-.527	-.214	-.841
Aspect	-.461	-.551	-.055	-.627	-.411	.118	-.588	-.364	-.275
Slope	.556	.344	-.563	.476	-.047	-.656	.417	.538	-.334
Elevation	.410	.208	.367	.655	.321	.343	.646	.027	.388
Litter	-.205	-.235	.627	-.233	.034	.572	-.201	-.372	.268
SLP IV							.455	-.652	.299

CCA = canonical correspondence analysis.
SLP = Shortleaf pine.

Table 3—CCA percent of variance in species data explained by axes

	CCA percent of variance explained			
	Axis 1	Axis 2	Axis 3	Cumulative
All species	15.9	7.5	5.7	29.1
Without SLP	15.7	9.2	8.1	33.1
SLP as environmental variance	15.4	11.1	8.7	35.2

CCA = canonical correspondence analysis.
SLP = shortleaf pine.

The percentage of variance in the data explained by the first three axes increases from 29.1 percent for the all-species ordination to 35.2 percent when pine is an environmental variable (table 3). This increase is largely due to the contribution of the second axis, which is most closely correlated with shortleaf pine importance. The proportion of variance explained by soils is displaced to the third axis.

DCA

DCA ordination of sample sites using all species is shown in figure 4. Patch types are again separated along the first axis. (Note that the orientation of the first axis is reversed from the CCA graph.) Riparian plots are grouped at the right end of first axis and in the middle range of the second axis; plantation plots form a group in the lower left quadrant. Forest plots are clustered in the center of the ordination space, except for outlier sites 3, 17, and 6, indicating unusual species compositions for these 3 sites.

Table 4 lists the correlation coefficients for the environmental variables and the distribution of sites in the ordination space. Height above the stream correlates most closely with the first axis (r = -.649). Consequently, the riparian sites (sites 10-15) and the upper sites on southwest facing slopes (sites 3, 4) occupy opposite ends of axis one. The second axis for the all-species ordination is most strongly correlated with slope (r = -.486) and organic litter (r-.411).

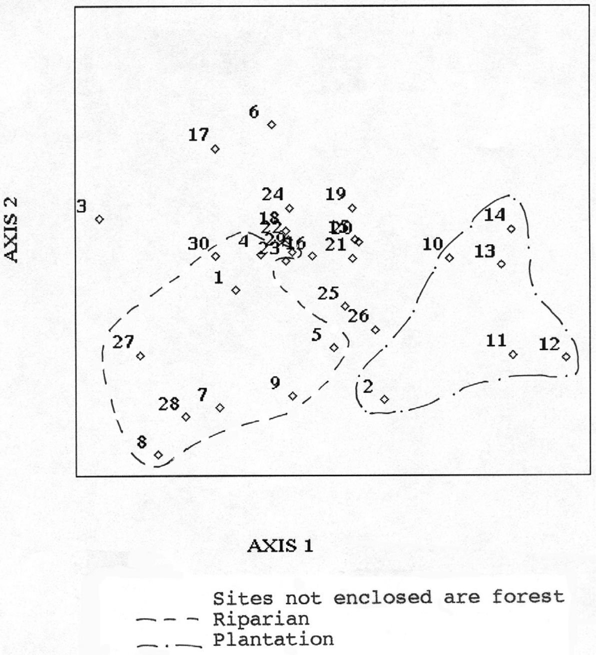

Figure 4—Detrended correspondence analysis ordination.

Table 4—Correlation of environmental variables with DCA axes (Pearson's r)

	Axis 1	Axis 2	Axis 3
Height above stream	-.649	.185	.058
Soils	.217	.084	-.251
Aspect	.356	.247	-.062
Slope	-.382	-.486	-.137
Elevation	-.353	.036	-.075
Litter	.094	.411	.009

DCA = detrended correspondence analysis.

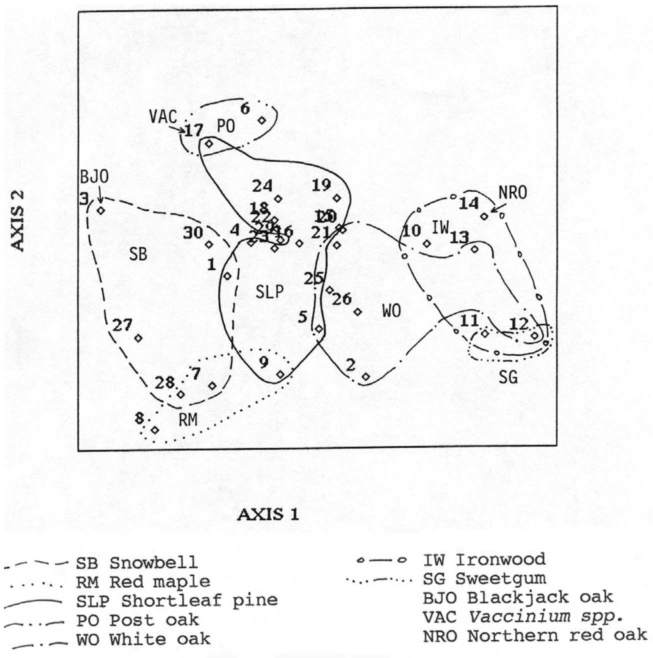

Figure 5—Detrended correspondence analysis ordination. Grouping of sites which exceed importance value of 15 percent for given species, or importance value of 50 percent for shortleaf pine.

Following the example of Ware and others (1992), sites with importance values ≥15 percent for given species are grouped in the ordination graph in figure 5. Since shortleaf pine exceeds 15 percent on all sites except riparian plot 12, sites with ≥ 50 percent pine are grouped. Five species other than shortleaf pine reach importance values ≥15 percent in two or more sites: Post oak (*Quercus Stellata* Wangenh.), snowbell (*Styrax grandifolius* Ait.), red maple, white oak (*Quercus alba* L.), and ironwood. There is little overlap of these groups. Only four sites have importance values ≥15 percent for other species when importance values for shortleaf pine are ≥ 50 percent: forest sites 17 and 1; and plantation sites 9 and 5. Only three of the sites with shortleaf pine importance value ≥ 50 percent are not forest; these are all plantation sites. The groups of secondary competitors are located at the periphery of the cluster of sites, with the shortleaf pine cluster in the center of the ordination space.

DISCUSSION AND CONCLUSION

The anomalous positions of site three in the DCA ordinations, and sites three, four, and six in the CCA ordinations deserve fuller discussion. Plots located at extreme positions in the ordination space frequently represent rare species occupying extreme sites (ter Braak 1987). Plot six is the only site located within the soil unit with the highest T factor. It has other unusual characteristics as well: it has the highest importance values for post oak (17 percent) and hickory (9 percent), and it is on a ridge top at 900 feet elevation. Only three sample plots are higher in elevation, and the two highest of these plots, three and four, are also outliers. Sites three and four are the two sites on the soil factor with the lowest T factor, and the only two on exposed southwest slopes. Plot three has an unusually high importance value (31 percent) for blackjack oak (*Q. marilandica* Muench.). Site four has an extremely high importance value for shortleaf pine (84 percent). CCA puts these unusual species compositions and unusual environmental conditions in extreme locations in ordination space.

Gradients

The most important gradient influencing species composition in this watershed is a moisture gradient, represented by height above the stream. As competition for light represented by shortleaf pine importance value is associated with the second CCA axis when it is used as an environmental variable, it is concluded that competition is the second greatest influence on the pattern of species distribution. Improved proportion of variance explained and improved correlation coefficients when pine is removed from the list of species variables or added to the environmental variables also support that conclusion.

The effect of competition on the pattern of species distribution is reflected in a comparison of the three CCA ordination diagrams (fig. 4). There is a marked separation of species response curves when a shortleaf pine importance gradient is introduced. The shift in site scores in the plantation patch (fig. 4C), where the planting of pines was fairly uniform is interesting. The orientation of the spread nearly parallel to the shortleaf pine importance vector may be explained by management practices, including uniform planting, that have been designed to foster the growth of shortleaf pine and suppress competition by other woody species. It indicates the relative success of early successional species. It must be noted, however, that the patch's elongation is largely due to the location of site 4, one of the sites with extreme environmental conditions. High shortleaf pine dominance at this site may be more directly the result of environmental stress that limits the success of other species rather than the effect of shading.

Centrifugal Organization

The pattern displayed in ordination space when species are grouped by importance values is consistent with centrifugal organization as proposed by Keddy and MacLellan (1990) (figs. 1 and 5). The dominant species, shortleaf pine, is in the center of the ordination space, while competitor species are grouped in specific locations around the periphery. In interpreting this distribution in can be inferred that the more peripheral sites represent sites with more extreme environmental conditions. The limited overlap among sites with ≥ 50 percent importance value of shortleaf pine and sites with ≥ 15 percent importance value of other species is evidence that shortleaf pine dominance inhibits the productivity of other species where competition with the dominant species is great. The clustering of the most successful species other than pine in specific locations around the periphery of the ordination space indicates that these species can achieve greater productivity in more extreme environmental conditions where adaptations to the specific conditions may give them a competitive edge.

Time

The Toots Creek ordination diagrams represent species distribution at one point in time—1996. The pattern is expected to change with time. The effects of disturbance or lack of it, human intervention, and time must be taken into account when interpreting the meaning of the pattern shown in these ordination diagrams.

Ordinations of this watershed in 30 years may show white oak displacing shortleaf pine from the center of ordination space. In forest ecology the dominant species in the optimal habitats are generally shade-tolerant. That is, they are the best competitors for other plant resources, soil moisture and nutrients, in low light conditions. In disturbance-free sites these species are expected to eventually overtop pioneer or intermediate sequential species. The central location of white oak in the ordination space suggests that white oak has this competitive strategy. When the shortleaf pine reaches the end of its relatively short life span, the white oak will be in a position in the shared optimal habitat to respond to increases in plant resources.

Keddy and MacLellan's centrifugal organization model assumes a natural species distribution in response to natural gradients. If management to maximize the productivity of the commercial species is successful, in 30 years one would expect an ordination of the plantation patch to show uniformity of composition, few species other than shortleaf pine, and little or no response to environmental gradients. This pattern represents the artificially induced species composition which is the objective of human intervention. As this pattern is not a result of naturally occurring variation, application of the centrifugal model is inappropriate.

Management Implications

One might feel justified in concluding that managed stands on all optimal sites do not reduce diversity, since beta diversity is greater in poorer site conditions. However, species richness is greatest (p = .05) in this watershed in the 80-year-old natural forest patch, currently occupying the optimal habitat. Furthermore, in these low stress environments, where all species could thrive were it not for competition, it is reasonable to conclude that periodic natural disturbance such as fire may result in the greatest species diversity. Forest ecosystem management that uses all optimal environments for plantations may then forfeit the greatest potential for increasing and sustaining biodiversity.

ACKNOWLEDGMENTS

Data for this study were collected by Gordon McWhirter, Hydrologic Technician; Clifford Harwell, Hydrologic Technician; and Dennis Carlson, Forester; put in usable form by Gail Henderson, Computer Specialist; and made available to me by Dan Marion, Research Hydrologist, USDA Forest Service, Southern Research Station, Oxford, MS. Hunter Speed, Silviculturalist, USDA Forest Service, Ouachita National Forest, Jessieville, AR, researched the management history of the Toots Creek watershed and Steve Brewer, Assistant Professor, Department of Biology, University of Mississippi, gave valuable criticism and insight.

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Session 8
Phase III Hydrology and Aquatic Ecology Research

Duane Dipert, Moderator

AN IMPROVED FLOOD-FREQUENCY MODEL FOR SMALL WATERSHEDS IN THE UPPER OUACHITA MOUNTAINS

Daniel A. Marion[1]

Abstract—A new regional flood-frequency (RFF) model is proposed for small, steep watersheds in the upper Ouachita Mountains of Arkansas. It is derived using Dalrymple's (1960) method and data from 10 monitoring stations with record periods of 15 to 33 years. First, I developed a preliminary RFF model and tested it against Neely's (1987) model using data from just five stations. Next, I compared the preliminary model's prediction accuracy to the accuracy of Neely's model using data from five stations not used to derive the new model. The preliminary model produced more accurate predictions than Neely's for all basins tested. Moreover, it's prediction errors are unbiased whereas those using Neely's model vary with recurrence interval and basin size. I derived a final RFF model using data from 9 of the 10 stations. It is applicable to sites within steep, forested basins of < 100 acres.

INTRODUCTION

Accurate predictions of peak streamflow magnitude are essential for accomplishing forest ecosystem management. Such predictions are required to design adequate drainage and stream crossing structures. They are needed to assess the risk of failure for in-channel or flood-plain structures, and for planning research and restoration projects. Peak flow predictions are made using flood-frequency (FF) models that relate discharge magnitude to the recurrence interval of such events (t_r, the average number of years between events), or its inverse, the probability of such a flow occurring in a given year ($P = 1/t_r$).

A series of equations have been defined by Neely (1987) for predicting peak discharge at given recurrence intervals (Q_r) throughout Arkansas.[2] These equations model Q_r as a function of various basin and climatic characteristics. They apply to all unregulated streams with drainage areas < 3,000 square miles within the entire State. Hereafter, the term "Neely's model" is used to refer in general to these equations.

Two reasons suggest that Neely's model may not be accurate in the smaller, headwater streams of the Ouachita Mountains. First, it is based primarily on catchments with relatively large basin areas. Within the Ouachita Mountains, only 3 of the 23 stations used by Neely have basin areas < 640 acres and only 2 are < 160 acres. Second, Neely's model is defined so as to minimize the influence of channel slope and elevation, two factors that are widely recognized as greatly influencing streamflow in small, mountain basins (Lee 1980). Neely states the latter is done to eliminate "bias" from these variables, but it's not clear from his discussion what exactly is meant by "bias."

In this paper, I present an improved FF model that is applicable to the upper Ouachita Mountain region (see figure 1). This new model is derived using the regional flood-frequency (RFF) method of Dalrymple (1960) and data from 10 small, steep watersheds. The presentation is organized as follows: first, Neely's model and procedures for deriving a preliminary RFF model using a data subset are described; second, Q_r predictions from the preliminary RFF model are compared to those using Neely's model; and last, the final RFF model

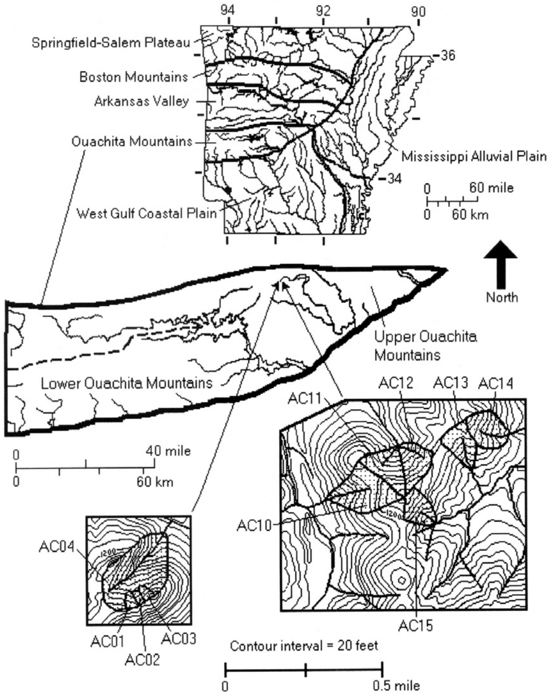

Figure 1—Physiographic provinces in Arkansas (Hodge and Tasker 1995); location of theu Ouachita Mountains; and gauging stations used to derive or test a regional flood-frequency model. Topographic data from Nimrod SE and Paron SW USGS 7.5-minute quadrangles.

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[2] Hodge and Tasker (1995) recently published an updated series of models for Arkansas. I discovered this work too late to include it within this paper, however, I did have time to test their models using the same procedures and found them to actually be less accurate than those of Neely (1987) for small, steep basins in the upper Ouachita Mountains.

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is derived using all appropriate data and its usage considerations are discussed. The region of interest is limited to the upper Ouachita Mountains where data are sufficient for this analysis.

METHODS AND DATA

Overview of Existing and New Models

Neely (1987) derived models for predicting 2-, 5-, 10-, 25-, 50-, and 100-year peak flows. He divided the State into two regions wherein different equations are applicable. The upper Ouachita Mountains region is entirely within Neely's Region B. Using multiple regression, he found that basin area, channel slope, mean annual precipitation, and mean basin elevation were the important factors in predicting Q_r. For small mountain streams like those in the upper Ouachita Mountains, slope and elevation are treated as constants and set equal to 30 feet per mile and 500 feet, respectively.

The RFF method is relatively simple to apply and is used to derive an alternative model for predicting Q_r. Its only assumption is that all individual basins within the region have streamflows with the similar distribution characteristics (Dunne and Leopold 1978). The method actually produces two related models: one for predicting Q_r given the mean annual flood (Q_{maf}) for a basin and the t_r of interest, and a second for predicting the Q_{maf} from basin characteristics. Hereafter, the term "RFF model" is used to refer to both models.

The procedures used in deriving a RFF model are explained in detail in Dalrymple (1960). Briefly, they are

1. Determine the candidate stations and base period. The base period is the time period that provides the greatest overlap in record periods between stations
2. Derive a FF model for each station
3. Predict the Q_{maf} for each station. The Q_{maf} is used as a scaling factor to remove the effect of basin size from the discharge data, so that a single model can predict flows for basins of varying size. It is determined for each station using the respective FF model and the discharge at the 2.33-year recurrence interval
4. Test the homogeneity of data from all stations. This test ensures that all stations have reasonably similar variability in peak discharges for their respective record periods
5. Derive the regional Q_r/Q_{maf} model. This model expresses the change in Q_r/Q_{maf} with t_r. It is derived by predicting the discharges at selected t_r values using the individual FF models, dividing them by their respective Q_{maf} values, determining the median Q_r/Q_{maf} value for all stations at each t_r, and modeling the medians against the t_r
6. Derive the Q_{maf} model. This model predicts Q_{maf} values so that the Q_r can be determined from the Q_r/Q_{maf} model at ungauged sites

Two exceptions are made to Dalrymple's procedures. First, both the Q_r/Q_{maf} ratio and Q_{maf} models are derived using regression procedures rather than fit-by-eye curves. Nonlinear regression is used to model Q_r/Q_{maf} versus return period, while linear regression is used to model Q_{maf}. Second, several basin characteristics are used to model Q_{maf} rather than just basin area.

Model Derivation and Testing Procedures

Data from 10 gauging stations located within the region (see figure 1) were used to develop and test the RFF model.[3] These stations are all located in small, steep basins. None of the other stations in the region were used because either they have drainage areas that are too large or have catchments that are too low in relief (see Neely 1987 for these data). The 10 stations used have basin areas from about 1 to 32 acres (see table 1). Their base record period is a little over 30 years, which is quite long for stations within such small basins. The data constitute annual maximum instantaneous discharges during the record periods for each station.

The 10 watersheds all have continuous, pine-hardwood forest cover, but vary in stem density, biomass, and overstory age as a result of past logging (see table 1). In this way, they are representative of small basins throughout the upper Ouachita Mountains where forest management has produced a mosaic of forest conditions.

The data were divided into two groups: one for developing a preliminary RFF model and a second for testing its accuracy against predictions using Neely's model. The model group includes five stations, four of which were chosen at random. One station, AC04, purposely was included in the model group because its basin area is substantially larger than all other stations (see table 1). This ensures that the preliminary model represents the range of conditions found among all 10 stations. The testing group includes the remaining five stations. Both groups include data from basins that cover the range of forest conditions found among the 10 stations.

Data for developing the FF models for each station were taken from the same base period of 1961–93. Where data are missing, they were estimated by finding the station among the other nine that was most highly correlated to the station with the missing data, regressing the discharges from those years when both stations have data, then estimating the missing data using the resulting model. Missing data estimates were used only in determining the ranks of valid data; they were not used in computing the FF models.

FF models were derived for each station in both the model and test groups. Nonlinear regression is used to fit one of two exponential decay models, either

$$Q_r = b_1 e^{-b_2 P} \qquad (1)$$

or

$$Q_r = b_0 + b_1 e^{-b_2 P} \qquad (2)$$

where

b_0, b_1, and b_2 = regression coefficients.

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[3] Unpublished data on file at the USDA Forest Service, Southern Research Station, P.O. Box 1270, Hot Springs, AR 71902.

Table 1—Environmental characteristics of Forest Service gauging stations used in the regional flood-frequency model development and testing

Station	Basin area	Basin relief	Forest conditions	Mean annual flood	Data group
	acres	*feet*		*cfs*	
AC01	1.63	75	Undisturbed	1.34	Test
AC02	1.28	75	Shelterwood harvest in 1970 and 1977, thinned in 1981	1.16	Test
AC03	1.44	60	Clearcut 1970, replanted in 1976	2.04	Model
AC04	32.50	220	Undisturbed	19.94	Model
AC10	14.17	220	Selection harvest in 1980 and 1987	13.14	Test
AC11	12.18	180	Undisturbed	9.00	Model
AC12	14.60	180	Clearcut, burned, and replanted in 1980	13.91	Model
AC13	11.71	180	Undisturbed	9.78	Model
AC14	10.76	140	Selection harvest in 1980 and 1987	11.14	Test
AC15	12.64	150	Clearcut, burned, and replanted in 1980	11.86	Test

Note: Record periods are 1961–93 for stations AC01-AC04 and 1979–93 for stations AC10-AC15.

This allows the FF models to be expressed as equations rather than graphs, which greatly facilitates their use. The models were computed using SigmaPlot and derived using the Marquardt-Levenberg algorithm (SPSS 1998).

Different modeling approaches and weighting schemes were used to derive a variety of candidate models for each station. Models were derived both by forcing b_0 to equal 0 and by computing b_0. Different weighting values were explored to de-emphasize the leverage of the largest peak flows in deriving the individual FF models. The largest discharges have the most uncertain probabilities and can have a disproportionate influence on model parameters. In contrast, the lower discharges have more certain probabilities, but have less influence because their residuals are typically small. Weighted regression can be used to minimize this effect (Myers 1990). Because this uncertainty varies with discharge (Q), then $1/Q$ and $1/Q^2$ were tested as weights, in addition to an unweighted model.

I chose the best model to be the one that maximizes overall model fit while conforming most closely to the higher discharge probabilities. I used the model R^2 to measure overall fit and visual inspection to determine quality of fit at the lower discharge values. When different models were equivalent using these criteria, I chose the simplest model (fewest parameters, least weighting).

The preliminary Q_r/Q_{maf} model was derived using the median values at 24 probabilities from 0.01 to 0.99. The probabilities were selected so as to adequately define the change in Q_r/Q_{maf} across the probability range for the five stations in the model group.

The preliminary Q_{maf} model was derived using linear regression to relate Q_{maf} to basin morphometry variables for all stations in the model group. Basin morphometrics considered were basin area, relief, length, and slope (see table 1). The best model was chosen considering R^2, model mean square error, and Mallow's C_p factor (Myers 1990).

Model accuracy for both the preliminary RFF and Neely's model was tested by comparing discharge predictions using these models against those predicted by the individual FF models for stations in the test group. My rationale was that because the individual models are based solely on the measured data at each station, they should provide the best estimate of the "true" Q_r at each station. This rationale is only as good as my assumption that the exponential decay functions used to fit individual models provide an accurate representation of annual peak discharge distribution. Because all selected models had R^2 terms of 0.92 to 0.97, I concluded this assumption was valid. Accuracy for both the preliminary RFF and Neely's model was evaluated qualitatively by visually determining which model predicts discharges closest to the individual FF predictions. It was assessed quantitatively by computing the root mean square (RMS) error of model predictions from corresponding station Q_r values. Hereafter, Q_r values predicted from individual station models are referred to as "station" values while those predicted using either the RFF or Neely's model are referred to as "predicted" values.

RESULTS AND DISCUSSION

Model Comparisons

The FF models for individual stations are fairly similar and all were fit with a high degree of accuracy, as noted above. An example is shown in figure 2. Best model fits were most often obtained using a two-parameter model ($b_0 = 0$) with $1/Q$ weighting. Stations AC03 and AC04 were best fit using the three-parameter model. Stations AC01 and AC10 were best fit using no weighting, while station AC03 was best fit using $1/Q^2$ weighting. With the exception of AC04, the two-parameter model derived using $1/Q$ weighting is almost as good as the selected model for these stations. The three-parameter model is clearly better for AC04. A possible—albeit unproven—physical interpretation of the need for the b_0 value is that due to its larger basin area (table 1), AC04 exhibits a minimum, positive discharge even at the lowest recurrence intervals, whereas the other, smaller basins approach 0 discharge.

Station AC10 is the only station that fails to pass the homogeneity test. Its variance exceeds the recommended limits (Dalrymple 1960); therefore this station was excluded in deriving both the preliminary and final RFF models. Why it differs from the other watersheds is not readily apparent. It is located adjacent to AC11-AC15 and has similar physical characteristics (see table 1).

The component models of the preliminary RFF are

$$\frac{Q_r}{Q_{maf}} = 2.237e^{(-1.885)P} \tag{3}$$

and

$$Q_{maf} = -0.317 + 0.423A + 0.031E \tag{4}$$

where

A = basin area (acres)

E = basin relief (feet).

The Q_r/Q_{maf} model is extremely well fit using a two-parameter function. This exceptional degree of accuracy is due to the smoothing effect produced by modeling only the median ratios at each probability. The best Q_{maf} model uses basin area and basin relief to predict Q_{maf}. Its R^2 is 0.95.

All comparison tests clearly show that the preliminary RFF model is more accurate than Neely's model for the basins considered. Plots of station versus predicted Q_r for three of the test group stations clearly show the difference (fig. 3). The stations used in figure 3 show the entire range in overpredictions and underpredictions using the preliminary RFF model. Surprisingly, the preliminary RFF model is most accurate for station AC10, the one station whose variance caused it to be excluded from the model group. The RMS errors confirm the visual evidence. The RMS error for the preliminary RFF varies between 0.6 and 4.7 cubic feet per second; while it varies from 4.7 to 27.9 cubic feet per second for predictions using Neely's model.

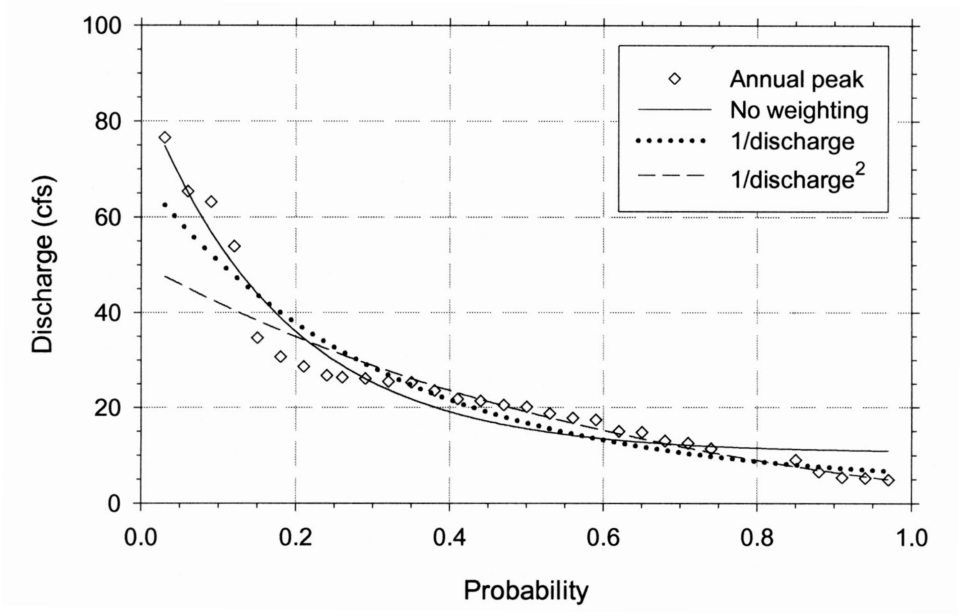

Figure 2—Flood-frequency models for watershed AC04. Different models are produced using nonlinear regression with the weighting methods listed. Scales are arithmetic and are the ones used in all regression analyses.

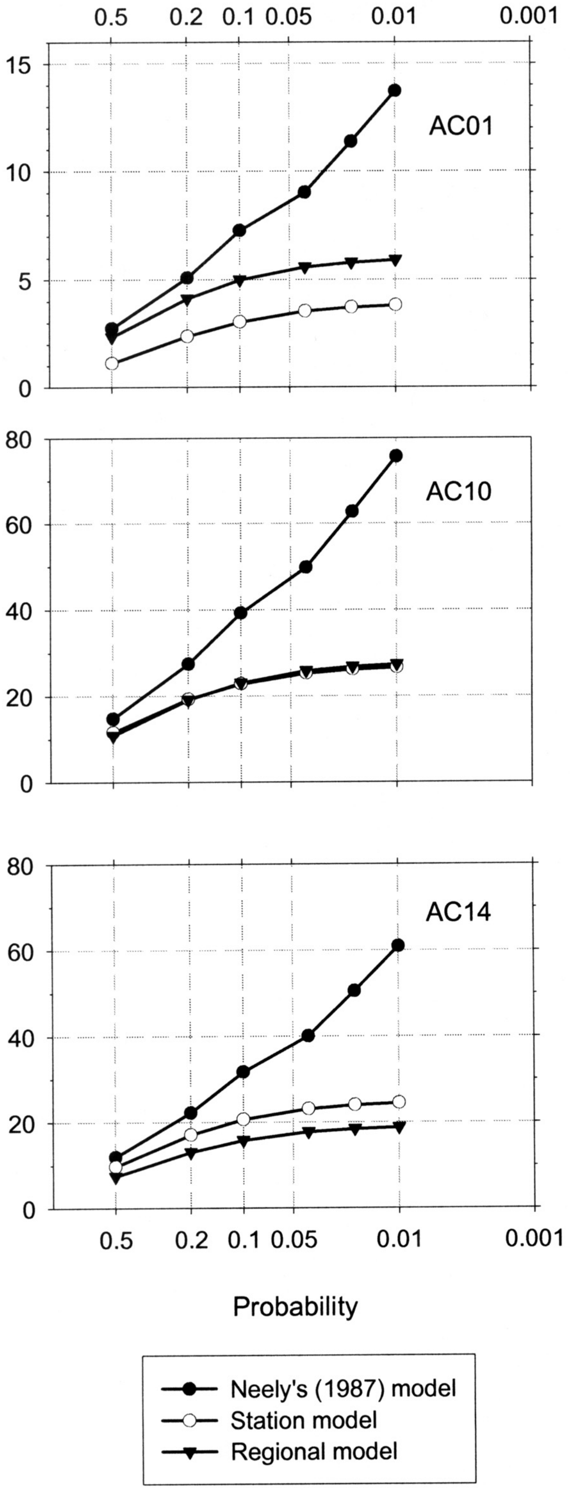

Figure 3—Comparison of discharge predictions using the preliminary regional and Neely's (1987) models. "Observed" discharges are predictions based on observed data for each station.

Discharge is consistently overpredicted using Neely's model and the error increases with both t_r and basin size. The distributions of prediction errors are shown in figure 4. Both the median error and the variance clearly increase in positive value (overprediction) as t_r increases; i.e., probability decreases, and basin size gets larger. Errors are especially pronounced at $P < 0.1$ (10-year flow) and area > 10 acres. In contrast, the preliminary RFF predictions show no bias with t_r and only minor fluctuations around 0 error with varying basin size.

Given the successful performance of the preliminary RFF model, a full model is derived using data from all stations except AC10. Its component Q_r/Q_{maf} and Q_{maf} models are

$$\frac{Q_r}{Q_{maf}} = 2.273e^{(-1.922)P} \tag{5}$$

and

$$Q_{maf} = -1.262 + 0.416A + 0.040E \tag{6}$$

The final Q_r/Q_{maf} model is shown in figure 5. Like the preliminary RFF, the selected best model for Q_{maf} uses basin area and relief. It has an R^2 of 0.94. Similarly, the Q_r/Q_{maf} model is fit with a two-parameter exponential decay function.

Model Application Considerations

Neely's (1987) model is not a *bad* model. The results presented above do not invalidate using Neely's model for sites where basin characteristics correspond more closely to the data used to develop that model. Given that Neely's objective was to provide a method for estimating Q_r throughout the entire state of Arkansas, it is not surprising that his model can be improved on in smaller areas like the upper Ouachita Mountains where additional data are available.

The final RFF model resulting from this work has limited applicability. Its accuracy outside of the upper Ouachita Mountain region is unknown. The model is based upon data from sites of approximately 1 to 35 acres and can be applied to similar basins within this size range. In addition, it has been my experience that basins with very similar relief and environmental characteristics occur up to about 100 acres in the upper Ouachita Mountains. Therefore, I think that where conditions are determined to be similar to those listed in table 1, the model should be applicable to catchments up to approximately 100 acres.

Past design and risk assessments based on Neely's model for sites in basins like those discussed here may still be valid. They may overpredict peak discharge, but factors other than discharge often need to be considered when designing in-channel structures or assessing their risk of failure. Such factors include the possibility that a structure like a culvert will become plugged with woody debris and leaves during high-flow events. Such factors typically act to increase the design flow that must be accommodated, thus overpredictions may be entirely appropriate.

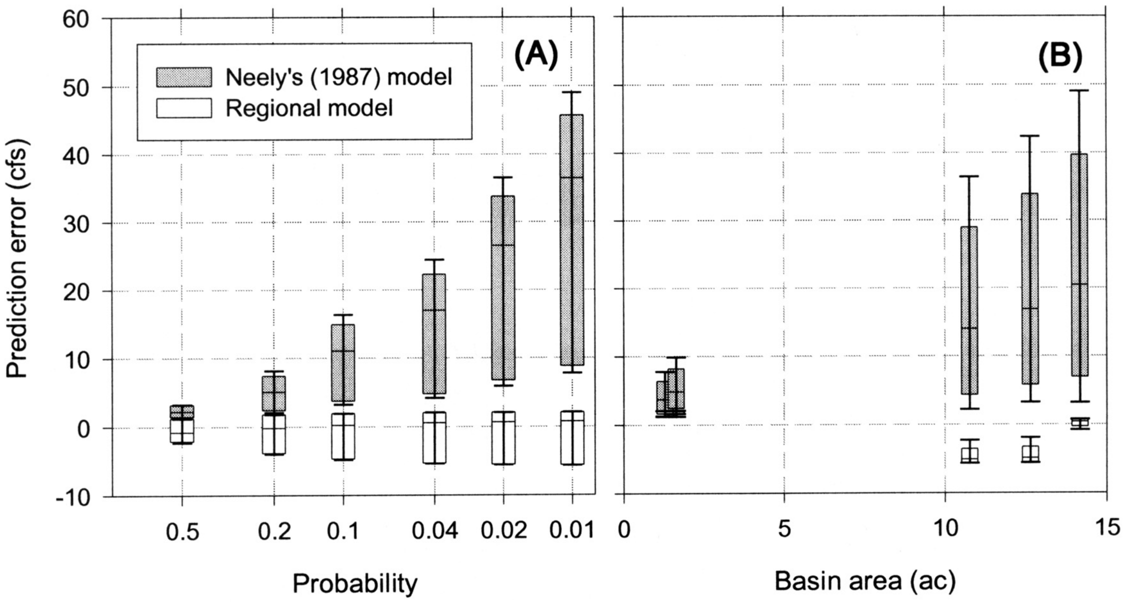

Figure 4—Box plots comparing the effects of (A) recurrence probability and (B) basin area on prediction errors using both the preliminary regional and Neely's (1987) models. Errors are plotted at six probabilities corresponding to 2-, 5-, 10-, 25-, 50-, and 100-year events, and at the basin areas for each of the five stations in the test group.

If a common frequency distribution for small-basin peak discharges can be determined, then a still better model might be developed. Such a model would allow more accurate prediction of extreme values, which are always the most uncertain estimates when using regression methods. It would also allow shorter record periods to be used in developing local models. However, identifying a common frequency distribution is difficult due to the short records generally available for small, mountain stations and the relatively large variability inherent in hydrologic processes at such fine spatial scales.

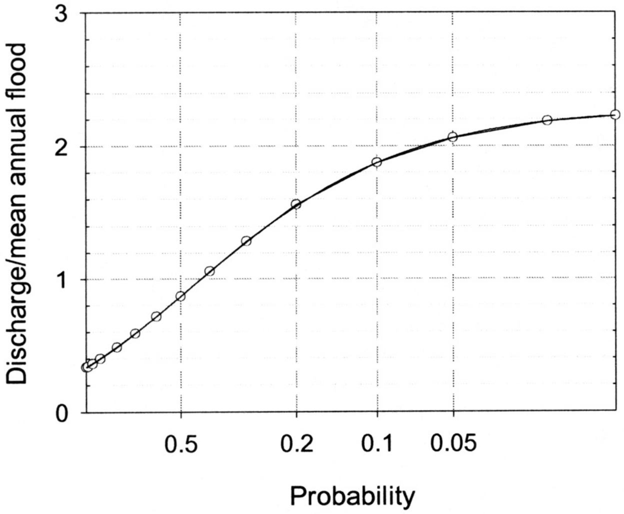

Figure 5—Final regional flood-frequency model for small, steep catchments in the upper Ouachita Mountains. For a given location, discharge for a given recurrence interval is predicted by calculating the Q_r/Q_{maf} ratio at the probability then multiplying this value by the Q_{maf} for the location. Q_{maf} is predicted from the second model. Both models are based on data from nine stations.

SUMMARY AND CONCLUSIONS

For small, steep watersheds in the upper Ouachita Mountains, the RFF model described here more accurately predicts Q_r than Neely's (1987) model. Neely's model overpredicts discharge for such basins. Using the RFF model requires measurement of the basin area and relief for the site in question. The RFF can be applied with confidence to watersheds of 1 to 35 acres, and probably can be applied up to 100 acres if environmental characteristics remain similar to those listed in table 1. Model accuracy is unknown outside of these limits where, if appropriate, previously developed models should be used. Factors in addition to discharge should be considered in designing and assessing failure potential for in-channel structures in headwater basins.

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NUTRIENT UPTAKE AND COMMUNITY METABOLISM IN STREAMS DRAINING HARVESTED AND OLD-GROWTH WATERSHEDS: A PRELIMINARY ASSESSMENT

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Abstract—The effect of timber harvesting on streams is assessed using two measures of ecosystem function: nutrient spiraling and community metabolism. This research is being conducted in streams of the southern Appalachian Mountains of North Carolina, the Ouachita Mountains of Arkansas, the Cascade Mountains of Oregon, and the redwood forests of northern California, in order to understand similarities and differences among stream ecosystem responses to timber harvesting across diverse geographic regions. Data from Cedar and Peacock Creeks in the redwood forest are used to illustrate the principles and usefulness of measuring stream ecosystem function for assessing watershed disturbances. Streams draining logged watersheds had smaller dominant substrate size and more sand and fine sediments in the channel. Nutrient uptake (NH_4^{+1}, PO_4^{-3}) and community metabolism (primary productivity, respiration, P:R, net daily metabolism) were measured in streams draining old-growth (Cedar Creek) and harvested (Peacock Creek) watersheds. Phosphate uptake length was significantly shorter in Cedar Creek (212 m) than in Peacock Creek (1020 m). Ammonium uptake length was not significantly different in these streams (111 m vs. 109 m, respectively). Preliminary analyses of stream metabolism suggest that primary productivity is greater in streams draining logged watersheds, but community respiration is greater in stream draining old-growth watersheds, resulting in sustantial differences in P:R and NDM.

INTRODUCTION

The Frontier Forest Initiative of the World Resources Institute reported that north temperate regions contain more than 50 percent of the world's healthy, intact "frontier" forests (World Resources Institute 1997). The United States has about 12 percent of the world's temperate forest and is harvesting these forests at a rate of more than 500,000 ha/y (World Resources Institute 1997).

The regenerated forests of temperate regions are often quite different from the original forests. Studies of the reforested areas of the northeastern United States have shown changes in tree species composition and smaller tree sizes compared with historical forests. Forest plantations may compensate for the loss of forest resources, but there is little evidence suggesting that they maintain pre-harvest levels of ecosystem function (Franklin 1988, Swank and Crossley 1988).

Timber harvests affected streams in most forested areas of the United States (Cummins 1980, Webster and Swank 1985, Webster and others 1992). van der Leeden and others (1990) reported that 7.5 percent of the total U.S. stream length has been impacted by logging and silvicultural practices.

Ecosystems are complex, self-regulating, functional units. Unlike communities and populations which are structurally defined, ecosystems are defined by rates and processes, such as energy flow or material cycling. These processes are mediated by the trophic structure of the ecosystem. Functional indicators are those metrics that directly or indirectly measure energy flow and material cycling within ecosystems. Functional indicators (such as ecosystem metabolism or nutrient cycling) are less likely to be constrained by regionally restricted biota. Thus, functional approaches lead to a more global view of stream ecosystems, a view that is much less variable than one based only on taxa inhabiting stream communities (Cummins 1988). Hunsaker and others (1990) stated that for regional ecological risk assessments to be effective, the system must be functionally defined, with the spatio-temporal boundaries of the system set by functional attributes of the communities inhabiting the system. Assessments that are functionally based are likely to have greater applicability across regions (Hunsaker and others 1990).

The most commonly measured functional attributes of ecosystems are primary productivity, or photosynthesis, (P) and community respiration (R). These two metrics are termed community metabolism when considered together. Two additional metrics are calculated from P and R: the ratio of P to and daily P-R which yields an estimate of net daily metabolism (NDM). These four metrics have been shown to be sensitive indicators of ecosystem stress (Bott and others 1985, Hill and others 1997). Metabolism is expected to be stimulated by nutrients and some organic substances and suppressed by some pesticides and organic substances, decreased pH, metals, and habitat degradation stress (Bott and others 1985, Hill and others 1997).

Many of the transformations which nutrients undergo while being transported into, through, and out of a stream segment are determined by difference between input and output. The actual pathway an atom of nutrient takes through this system, usually inferred from movements of radioactive tracers from one ecosystem component to another, is described as spiraling (Webster and Patten 1979).

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Elwood and others (1983) listed three aspects of the spiraling concept that make it useful for the analysis of stream ecosystems. First, it is a simple means of measuring nutrient (including carbon) flow through streams. Second, this concept emphasizes the interaction of hydrologic transport and ecological processes governing nutrient cycling. Third, the interaction of both spatial and temporal dynamics of streams is emphasized. Additionally, spiral length may be indicative of increases in nutrient turnover, transport, and loss from streams as predicted for stressed ecosystems.

Newbold and others (1981) developed an index of nutrient spiraling known as spiraling length, defined as the average downstream distance associated with one complete cycle of a nutrient atom. In developing this index, they assumed that a nutrient atom passes through water, particulate, and consumer compartments while being transported. These components of the spiral can be quantified and the sum of their average distances is the spiraling length.

Investigations of factors controlling spiral length have shown a greater role by the biotic component (e.g., Elwood and others 1981; Newbold and others 1982, 1983; Mulholland and others 1985b, 1985a) than by abiotic factors. Spiral length is decreased by those organisms that retain organic matter and nutrients within a stream (e.g., sessile microorganisms, filter-feeding insects, and grazers), and increased by organisms which promote particle transport (e.g., shredders). Elwood and others (1981) found that 91 percent of exchangeable phosphorus uptake by fine particulate organic matter (FPOM) was due to microorganisms. Similarly, about 80 percent of the exchangeable phosphorus uptake by stream sediments was due to microorganisms. These findings are in agreement with those of Gregory (1978) who reported less than 20 percent of the phosphorus uptake by stream sediments and organic matter was abiotic.

Newbold and others (1983) reported that uptake length accounted for nearly 90 percent of the total spiral length. Mulholland and others (1985b) found uptake length to be as much as 98 percent of spiral length. If spiral length can be estimated accurately (± 10 percent) from uptake length, then pulse additions of non-radioactive phosphorus and ensuing downstream depletion of this phosphorus should serve as a measure of phosphorus spiraling in those streams where the use of radioactive tracers would be undesirable (Stream Solute Workshop 1990, Mulholland and others 1990, Webster and others 1991). Uptake length is expected to increase with most stressors, though some organic substances may result in uptake length shortening.

The goal of this research is to evaluate stream ecosystem function in response to different timer harvesting intensities and time since harvesting. The research is being conducted in streams of the southern Appalachian Mountains of North Carolina, the Ouachita Mountains of Arkansas, the Cascade Mountains of Oregon, and the redwood forests of northern California. A secondary objective will be the assessment of intra- and inter-annual variability in measures of stream ecosystem function. We predict that any amount of watershed disturbance by logging will affect ecosystem function by depressing community metabolism and increasing nutrient uptake lengths.

METHODS

Study Sites

The study is being conducted at 24 sites grouped by different levels of watershed degradation due to the intensity of forestry management practice. Sites have been selected at National Science Foundation Long-Term Ecological Research (LTER) sites and US Forest Service (USFS) research units at the Southern Research Station and Redwood Sciences Laboratory. Sites are of comparable watershed size. The LTER sites have extensive historical databases on hydrology, chemistry and aquatic community structure and managed impact programs. The USFS sites have historical data on hydrology, physical habitat, aquatic community structure and timber harvest.

Coweeta Hydrologic Laboratory—The Coweeta Hydrologic Laboratory is a 2,185 ha experimental facility in North Carolina administered by the USFS. It has been dedicated to forest hydrology research since its establishment in 1933. Stream sampling has been conducted on eight mixed hardwood control areas and 13 catchments where forest management prescriptions have been applied.

Ouachita National Forest—The Ouachita National Forest is an 800,000 ha mixed hardwood-conifer forest in central Arkansas. Research sponsored by the USFS Southern Research Station has been conducted on streams in the unit since 1988. In 1990, clear-cutting was eliminated and a research program initiated to examine the effects of new forest management strategies on stream ecosystems.

H. J. Andrews Experimental Forest (LTER)—The H.J. Andrews Experimental Forest, located in the Cascade Range of Oregon, is a 6,400 ha drainage basin administered by the USFS Pacific Northwest Research Laboratory. When it was established in 1948, the basin was covered with virgin forest. Old-growth forest stands with dominant trees over 400 years old still cover about 40 percent of the total area. Mature stands (100 to 140 years old) originating from wildfire cover about 20 percent. About 30 percent of the Andrews Forest has been harvested, creating young plantation forests varying in composition and age.

Redwood Sciences Laboratory—Unlike the other study units, which have their experimental watersheds located in close proximity, the redwood forest study unit has widely dispersed experimental watersheds. These study watersheds are an aggregation of management units of the Six Rivers National Forest and the Redwoods State and National Park and are scattered over a large portion of California's northwestern coastal mountains. Research has been conducted for several years in these units by the USFS Redwood Sciences Laboratory and the U.S. Geological Survey.

At present, only data from the redwood forest study unit has been analyzed. For the purpose of demonstrating the potential use of community metabolism and nutrient spiraling, Cedar and Peacock Creeks in the Smith River drainage basin east of Crescent City were selected.

Nutrient Uptake/Spiraling

Phosphate and ammonium uptake lengths were determined using the method described by the Stream Solute Workshop (1990), Mulholland and others (1990) and Webster and others (1991). A solution containing known concentrations of K_2HPO_4, NH_4Cl, and NaCl tracer was added to the stream and downstream changes in concentration measured. The solution, containing 5-10 times background concentrations of NH_4^{+1} and PO_4^{-3}, was added at a uniform rate until a plateau in tracer concentration was detected at the most downstream station (fig. 1). Once plateau concentration was achieved, triplicate samples were collected from stations located every 10 m along the 100 m stream reach. Samples were refrigerated until analyzed for NH_4^{+1} and PO_4^{-3} (APHA 1998).

The rise, plateau and decline of solute added to the stream was monitored by recording Cl^- concentration and specific conductivity every 30 seconds for the duration of the nutrient addition. Data logging continued until the specific conductivity returned to pre-release levels. Transient storage (A_s), the temporary storage of solutes in water that is moving more slowly than the main stream flow, was estimated as the difference between plots of the actual and predicted rise and fall of the tracer solution and adjusted for total streambed area (Stream Solute Workshop 1990).

Measured concentrations of nutrients and tracer were corrected for background levels and adjusted for hydrologic dilution (based on reduction in chloride concentration). Downstream concentrations were divided by the most upstream concentration, and the log of these concentrations regressed against distance (Webster and others 1991). The slope of the regression yields an uptake rate per unit stream length and the inverse of this is uptake length (Newbold and others 1981).

Stream metabolism—Total stream metabolism was measured *in situ* using continuously recording dissolved oxygen meters deployed at a single, downstream point in each study reach (Owens 1974; Young and Huryn 1996, 1999). The meters recorded dissolved oxygen (DO), percent DO saturation, and temperature every 15 minutes for 24 hours. Reaeration was determined by regressing the rate of change in DO concentration against the oxygen deficit (saturation DO concentration less actual DO concentration) (Owens 1974). Daily primary productivity and respiration were estimated by plotting the reaeration adjusted DO concentrations against time. Mean night-time respiration was extrapolated to estimate day-time respiration. Primary productivity is the area above the respiration line, and respiration is the area below the line (Bott 1996).

RESULTS

Physico-Chemical Characteristics

The two redwood forest study streams were similar in many of their physical characteristics, with the exception of wetted width, riffle to pool ratio, dominant substrate, and the amount of sand in the channel. Cedar Creek had lower background PO_4^{-3} and NH_4^{+1}(14.6 ± 0.030 and 29.5 ± 2.4µg/L) than Peacock Creek (18.9 ± 0.106 and 63.0 ± 8.5µg/L) (table 1).

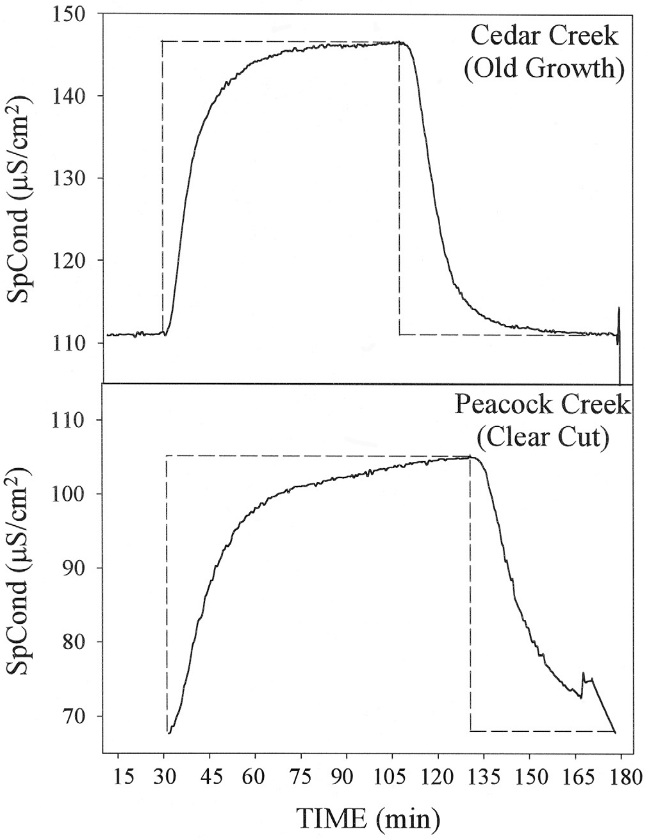

Figure 1—Actual (solid line) and predicted (dotted line) tracer concentration for solute releases in Cedar Creek (top) and Peacock Creek (bottom). The area differences between the two curves is an estimate of transient storage.

Nutrient Uptake/Spiraling

Three measures related to nutrient uptake and spiraling were considered in this preliminary study: transient storage, nutrient uptake rate, and nutrient uptake length. Transient storage in Cedar Creek (old growth) was 25 m^2 compared to 42 m^2 Peacock Creek (clear-cut) (table 2, fig. 1).

Phosphate uptake rate in Cedar Creek was 0.0066 µg/m compared to 0.0010 mg/m in Peacock Creek. These uptake rates translate into uptake lengths of 212 m and 1020 m, respectively (table 2, fig. 2).

Ammonium uptake rates in Cedar and Peacock Creeks were 0.0167 mg/m and 0.0138 mg/m, resulting in uptake lengths of 111 m and 109 m, respectively (table 2, fig. 3).

Community Metabolism

Four measures of community metabolism were measured in each of the study streams: gross primary productivity, respiration, P/R, and NDM. Gross primary productivity was 450 mg O_2/d in Cedar Creek (old growth) and 600 mg O_2/d in Peacock Creek. Respiration in these two streams was 690 mg O_2/d and 528 mg O_2/d. Net daily metabolism in Cedar Creek was -240 mg O_2/d compared to 72 mg O_2/d for Peacock Creek. The ratio of primary productivity to respiration (P:R) was 0.65 and 1.13 for Cedar and Peacock Creeks (table 2, fig. 4).

Table 1—Physico-chemical descriptions of Cedar Creek (old growth) and Peacock Creek (clearcut)

	Cedar Creek	Peacock Creek
Mean (± SE) channel width (m)	2.9 (0.3)	2.9 (0.2)
Mean (± SE) wetted width (m)	2.1 (0.2)	2.9 (0.3)
Streambed area (m^2)	286	291
Wetted width:channel width	0.73	1.00
Mean (± SE) depth (cm)	16 (3)	14 (2)
Maximum depth (cm)	67	46
Mean (± SE) canopy cover (%)	95 (1)	96 (1)
Mean (± SE) channel slope (°)	1.9 (0.4)	1.4 (0.4)
Riffle:pool	2.3	1.5
Dominant substrate	Cobble	Coarse gravel
Sand and fine sediments (%)	0	8
Mean temperature (°C)	13.2	12.9
Temperature range (°C)	12.6–13.5	12.8–13.1
Weather conditions	Foggy	Foggy
24-h antecedent precipitation (mm)	0	0
Background NH_4^{+1} (± SE) (µg/L)	29.5 ± 2.4	63.0 ± 8.5
Background PO_4^{-3} (± SE) (µg/L)	14.6 ± 0.03	18.9 ± 0.11
Harvest history	Never logged	2–3 clearcuts
Fire history	Unknown	Common
Watershed geology	Franciscan ultramafic	Franciscan ultramafic

Table 2—Measurements related to nutrient spiraling (transient storage, uptake rates, uptake lengths) and community metabolism (net primary productivity, community respiration) in Cedar Creek (old growth) and Peacock Creek (clearcut)

	Cedar Creek	Peacock Creek
Transient storage (m^2)	25	42
NH_4^{+1} uptake rate (mg m^{-1})	0.0167	0.0138
PO_4^{-3} uptake rate (mg m^{-1})	0.0066	0.0010
NH_4^{+1} uptake length (m)	111	109
PO_4^{-3} uptake length (m)	212	1020
Gross primary productivity (P)(mg O_2 d^{-1})	450	600
Community respiration (R) (mg O_2 d^{-1})	690	528
P:R	0.65	1.13
Net daily metabolism (NDM) (mg O_2 d^{-1})	-240	72

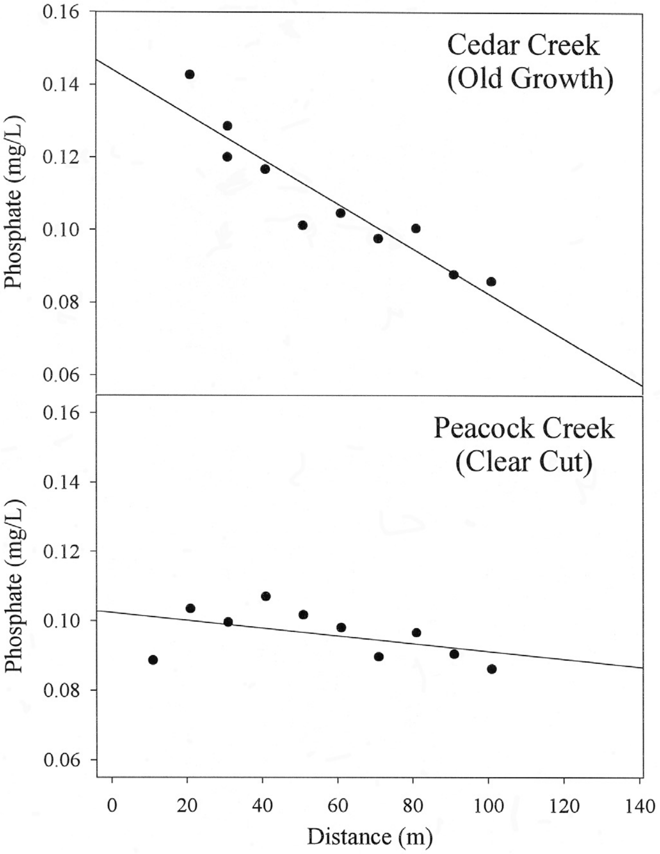

Figure 2—Regression of PO_4 concentration against downstream distance for Cedar Creek (top) and Peacock Creek (bottom). The slope of the regression line is the PO_4 uptake rate.

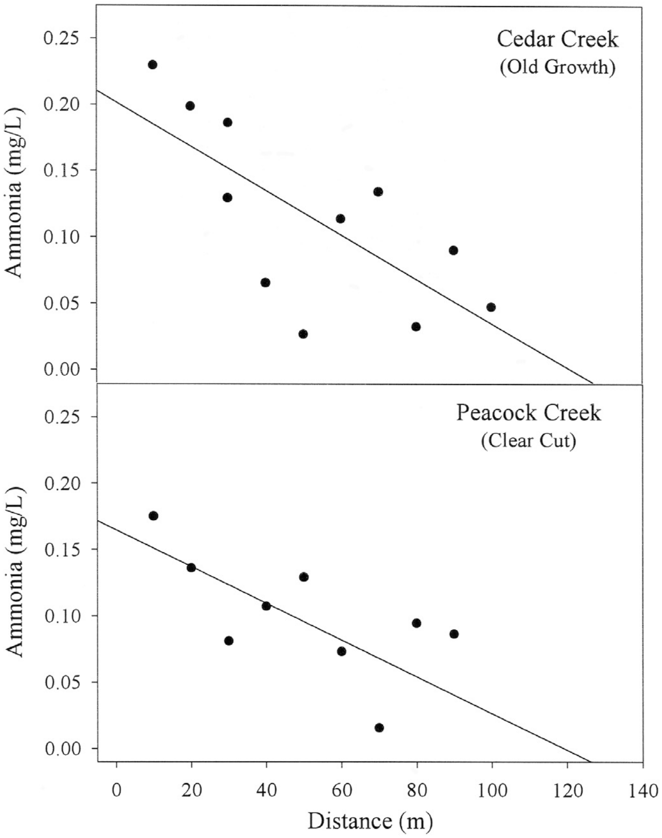

Figure 3—Regression of NH_4 concentration against downstream distance for Cedar Creek (top) and Peacock Creek (bottom). The slope of the regression line is the NH_4 uptake rate.

DISCUSSION

Logging activities, such as trail and road building, skidding, and slash removal, result in increased sedimentation in streams draining those watersheds (Lieberman and Hoover 1948, Golladay and others 1987, Webster and others 1992). Sediment influx to streams following logging is initially high, decreases slightly as the watershed begins revegetation, rises again during the transition from pioneer to later successional vegetation, then declines as the forest matures (Vitousek and Reiners 1975, Webster and others 1992).

Subtle differences in physical characteristics of Cedar and Peacock Creeks are probably a result of timber harvesting. Three measures, the ratio of wetted width to channel width, dominant in-stream substrate, and percent of the streambed as sand and fine material, were used as indicators of stream sedimentation. Wetted width:channel width in the clear-cut stream (Peacock Creek) is 1.00 indicating a lack of channel braiding and complexity. Substrate in Peacock Creek is dominated by smaller material (coarse gravel, 16-64 mm diameter) than that dominating in Cedar Creek (cobble, 64-250 mm diameter). Likewise, Peacock Creek had more sand and fine sediments in the channel than did Cedar Creek.

Increased nutrient export from logged watersheds has been demonstrated in several studies (Bormann and others 1974, Vitousek and Reiners 1975, Campbell and Doeg 1989, Webster and others 1992). Peacock Creek was exporting more nutrients than Cedar Creek, especially PO_4^{-3}.

The physico-chemical similarities between these two streams suggests that differences in nutrient uptake and community metabolism are likely due to differences in harvest histories.

Timber harvesting has been shown to affect nutrient processing in streams draining logged watersheds. Webster and others (1991, 1992) reported decreased PO_4^{-3} retention in logged streams, resulting in longer uptake lengths, and attributed this to biotic and abiotic changes in the stream. We found no differences between our two study streams in transient storage or NH_4^{+1} uptake, but PO_4^{-3} uptake in Peacock Creek was much lower, resulting in much longer uptake lengths.

Community metabolism in small streams draining forested watersheds tends to be dominated by processing of organic carbon derived from the watershed rather than generated *in situ* (Webster and others 1995), resulting in a P:R < 1 and a NDM < 0. As the riparian canopy opens as a result of logging, primary productivity increases (Webster and others 1983) and P:R approaches 1. Even with no difference in riparian canopy between Cedar and Peacock Creeks, net

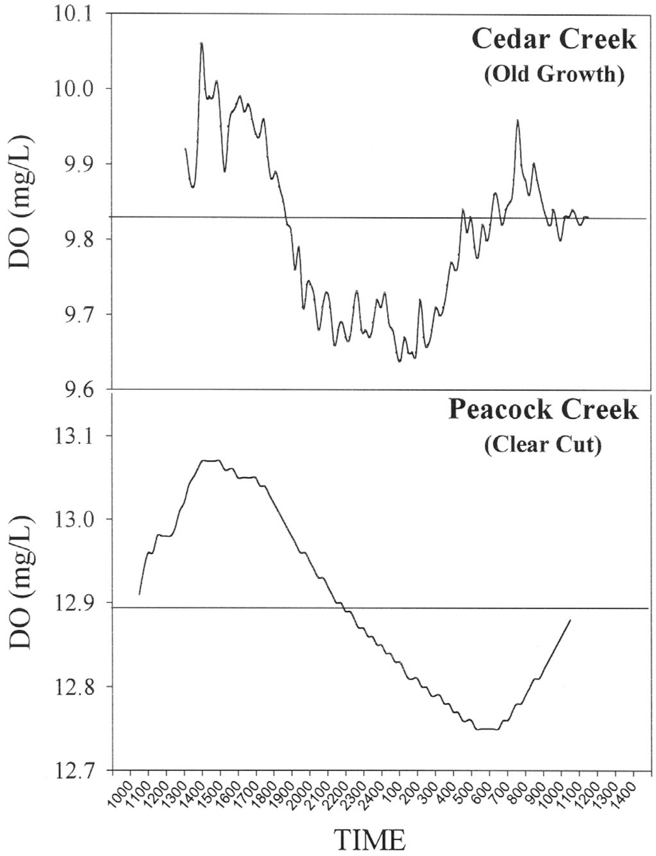

Figure 4—Diel dissolved oxygen curves for Cedar Creek (top) and Peacock Creek (bottom). Net primary productivity is the area under the curve and above the reference line. Community respiration is the area under the curve and below the reference line; DO = dissolved oxygen.

primary productivity was higher in Peacock Creek. This may be related to the higher nutrient levels in this stream. Community respiration, however, was lower in Peacock Creek, suggesting that the microbial community may have been carbon limited. Carbon limitation in streams draining logged watersheds is related to decreases in particulate organic carbon inputs, the depletion of organic carbon storage, and decreased dissolved organic carbon influx (Meyer and Tate 1983, Golladay and others 1989). Our results are consistent with the suggestion that disturbed watersheds are less efficient at processing organic carbon, as indicated by the lower respiration in Peacock Creek, which results in a positive NDM and a P:R > 1.

The preliminary results of our study of nutrient spiraling and community metabolism in paired old-growth and clear-cut watersheds demonstrates the potential use of these measures to assess the impacts of watershed disturbances on stream ecosystem function. While the patterns are clear, we must temper our interpretations of the data with the knowledge that our data at present are limited in number, unreplicated, and in every sense preliminary. Over the three-year term of this study, we will achieve the experimental replication and refinement of measurements of hyporheic storage and release of nutrients, stream physical characteristics, stream reaeration, and light regime, to achieve statistically robust inter-regional comparisons of the impacts of logging on stream ecosystem function.

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CONTEXT-SPECIFIC TROPHIC AND FUNCTIONAL ECOLOGY OF FISHES OF SMALL STREAM ECOSYSTEMS IN THE OUACHITA NATIONAL FOREST

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Abstract—Fish play diverse and important roles in stream ecosystems, but details about ecosystem effects are poorly known for many freshwater fish species. A requisite first step to understanding functional roles of individual species is information on their trophic ecology in the context of particular environmental settings. Stomach contents were analyzed for approximately 1,600 individuals of 21 common fish species (mostly insectivores or omnivores) from four streams in the Ouachita National Forest, Arkansas. Two streams (South Fork Alum Creek and Crooked Creek) were in watersheds with little recent timber harvest and two (Bread Creek and Blaylock Creek) were in watersheds harvested under Forest Service standards and guides. Individual fish species were distinctive in kinds of foods eaten, and were not substitutable for each other in ecosystems. Benthic and water column species differed in diet, but interspecific differences in food consumption were not closely linked to taxonomic similarity. Composition of fish diets were overall 70 to 85 percent similar between harvested and non-harvested watersheds, with dipterans and terrestrial insects the major foods used by all fish assemblages; most individual species had similar diets in different creeks or longitudinally within creeks. Foods eaten per species, multiplied by estimates of fish density from snorkeling and seining surveys, resulted in an order of magnitude prediction of consumption of about 13,000 invertebrate food items per day per 100 m of stream by the local fish assemblage.

INTRODUCTION

Small streams draining the Ouachita National Forest in Arkansas and Oklahoma are inhabited by a large number of native fishes with 117 species known from the Ouachita Highland (Matthews and Robison 1998). These fishes, dominated numerically by minnows (Cyprinidae), suckers (Catostomidae), catfishes (Ictaluridae), topminnows (Fundulidae), sunfishes (Centrarchidae), and darters (Percidae), can be very abundant locally. Thousands of individuals frequently occupy a single pool-riffle complex in these small upland streams. Matthews and Robison (1998) found a modal number of 13 fish species per collection across nearly 200 sites in the Ouachita Mountains, and collection of up to 20 species per site was not unusual. Until the 1980s, little emphasis was placed on the ecological roles of stream fishes, but it is now well known that feeding, nutrient release, and mechanical activities of fishes can have strong and diverse effects in stream ecosystems (Matthews 1998). Beginning with work by Power and Matthews (1983), Power and others (1985), and Grimm (1988), there is clear demonstration that some fish species have strong effects on structural and functional properties of North American stream ecosystems. Expanding the initial work of Power and others (1985), several papers (e.g., Gelwick and Matthews 1992, Gelwick and others 1997, Vaughn and others 1993) demonstrated that a single highly abundant, algivorous species [central stoneroller, *Campostoma anomalum* (Rafinesque)] had direct or indirect effects on at least 20 important structural or functional properties of small stream ecosystems (Matthews 1998). The central stoneroller can cause differences in algal community composition or productivity, uptake and dynamics of organic matter, invertebrate community composition or life history, movement of materials in streams, carbon-nitrogen ratios, and standing crops of bacteria. The species also plays a major role in predator-driven trophic cascades and can alter the manner in which stream ecosystems function overall. The central stoneroller is a highly abundant species in many streams of the Ouachita National Forest where it occurs in schools of up to several thousand and grazes algae vigorously and actively from rock substrates of flowing pools and riffles.

The demonstration that a single common species can change the dynamics of a stream ecosystem, while interesting, falls short of providing managers with information on the overall impacts of fish communities on stream ecosystems. The best assumption from a management view is probably that all fish species in a small stream have the potential to change or influence important dynamics in that stream, such as primary productivity, nutrient relationships, breakdown of particulate organic matter, linking individual pools with each other, or changing algal or invertebrate communities. To demonstrate this concept, (Gido and Matthews, in press), recently tested the hypothesis that a water-column minnow species (red shiner, *Cyprinella lutrensis* Baird and Girard) could have direct or indirect effects on benthic primary productivity. Over one fall and winter, they showed a direct positive relationship between the density of red shiners and primary productivity on artificial substrates in large outdoor experimental streams. However, the red shiners did not cause a decrease in density of benthic stream invertebrates (many of which graze algae). Presumably, the enhancement

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of benthic primary productivity resulted from an increased rate of nutrient transfer from surface or water column invertebrates, a primary food of red shiners, to nutrient limited periphytic algae. Regardless of the mechanism, (Gido and Matthews, in press) provided a clear demonstration that fishes other than strictly benthic species can have major impacts on stream ecosystem processes, including processes in the benthic (stream bed) compartment.

Regionally, good information about ecosystem effects is available for only one species (*Campostoma anomalum*) that is common in Ouachita National Forest streams, out of the more than 200 fish species recorded from Oklahoma and Arkansas. The long-range goal of our research program is to determine effects of a wide range of fish species on ecosystems of small streams, adopting the functional groups approach outlined by Matthews (1998).

The requisite first step to any assessment of the functional effects of a fish species on stream ecosystems is acquiring detailed, context specific information on its food use. That is, what (and how) does a fish eat and what is the specificity or plasticity of its feeding? Does a given species use largely the same food items regardless of its location in a stream, the stream system in which it occurs, its potential competitors for food, or availability of foods? Additionally, there is great need to understand if a particular fish species uses foods most like those of closely related species, and the extent to which species are redundant, or "substitutable" for one another, in an ecosystem. If a fish species feeds consistently regardless of environmental context, broad a priori predictions of its effect in an ecosystem should be possible. If feeding (and functional role) of a fish is more context-specific, focus would shift to more local, site-specific studies of fish communities in order to understand ecosystem dynamics. Managers should bear in mind that the fish community and the ecosystem are intricately linked in two "directions." It is becoming clear that fish have effects in ecosystems, but it is equally clear that the condition of the ecosystem plays a major role in the kinds and abundances of fish the system can support. Thus, the fish-ecosystem relationship can be considered a feedback loop in which the ecosystem affects the fish and the fish affect the ecosystem. Only by understanding effects of anthropogenic activities on the overall ecosystem, and this link to fishes, can an appreciation be gained of the interlinked roles of man, fish, invertebrates, and stream quality that actually comprises the stream ecosystem at the watershed level.

This study describes a first approximation of the use of foods by a substantial number of stream fish species of the Ouachita National Forest. Food use is examined in the context of stream, season, timber harvest regime, and, for selected species, longitudinal position in the stream. Specifically, general food use is summarized for 21 species in each of two pairs of streams in the upper Little Missouri River drainage and the upper Saline River drainage. Further, similarities and differences in food use are tested among species, two major feeding groups, and taxonomic relatives. Assemblages are examined from the perspective of food use differences among streams and timber harvest regimes. For selected species, intraspecific differences in food use are examined among or within streams and between harvest regimes. Finally, an order of magnitude estimate is calculated to ascertain the potential effects of fish assemblages on the stream invertebrate community.

METHODS AND MATERIALS

Study Area

Stream fish communities were sampled by seining at 24 sites on four dates (November 1989, May 1990, October 1990, May 1991). Six sites each were located in Blaylock and Crooked Creeks (Little Missouri River drainage, northwest of Glenwood, AR), and six each in South Fork Alum and Bread Creeks (Saline River drainage, north of Hot Springs, AR). All sites were on public lands of the Ouachita National Forest (Matthews 1993). Within each of the two river drainages, one creek was considered "harvested" and one, "nonharvested" on the basis of recency of timber harvest within the watershed (Clingenpeel 1994) and unpublished basin area stream survey data (Alan Clingenpeel, Forest Hydrologist, Ouachita National Forest, Box 1270, Hot Springs, AR). At the time of field sampling, about 12 percent of the stands in Blaylock Creek had been harvested in the previous 20 yr, whereas essentially no stands had been harvested in Crooked Creek in that time period. Similarly, about 21 percent of the stands in the Bread Creek watershed had been harvested in the last 20 yr, and no stands had been harvested in South Fork Alum Creek watershed during that period.

At each site, fish were sampled by seining with small seines in all identifiable habitats (e.g., pools, riffles, runs) for approximately 1 to 1.5 hr over about a 200-m stream reach. Each site included multiple pools and riffles, although some of the extreme headwaters sites (particularly in Bread Creek) were sometimes reduced to a series of disconnected pools. All fish were preserved in the field in 10 percent formalin, transported to the University of Oklahoma Biological Station, and later were identified, enumerated, and transferred to 50 percent isopropanol for permanent storage. A summary of the fish collected is provided in Matthews (1993).

Subsequent to fish sampling, the Ouachita Mountain Ecosystem Management Research Project (Southern Research Station) provided funding for assessment of stomach contents of the fishes collected at the 12 sites. Matthews, Miller, and Stewart revisited all study streams and conducted sampling to make a reference collection of all possible benthic invertebrates from the study streams. Invertebrates were sampled qualitatively (i.e., dipnetting, kicknetting, and picking invertebrates from stones). These samples were identified and enumerated primarily by Miller, who holds a Masters of Science degree on distribution and ecology of benthic stream invertebrates. She was assisted in identifications by Dr. David Bass, Central Oklahoma University, an authority on benthic stream invertebrates in the region. Thus, fish stomach content was assessed against the background of a detailed reference collection of invertebrates from the study streams. This allowed creation of a templet for recording contents of individual stomachs that included 65 invertebrate taxa arranged hierarchically so that results could be viewed at the lowest taxonomic categories or aggregated by family, order, etc.

Miller, Stewart, and other technicians (directed by Miller or Matthews) conducted examination of stomach contents of over 2,000 individual fishes. Individual fishes were chosen from across the original field collections and in approximate proportion to the abundance of the fish species in the field (table 1). Each fish was weighed and measured, and eviscerated. The stomach was weighed, opened and thoroughly cleaned of contents, and then re-weighed to obtain weight of total foods. Stomach contents were identified using binocular dissecting microscopes. Eviscerated fish were returned to archival storage at the Biological Station. Data on counts of food items were coded by hand by Miller and others, and the data sheets sent to Warren who directed entry of the data into a relational database. This database was ultimately returned to Cobb who carried out all data manipulations in a spreadsheet format. In the present study, all data were aggregated at the level of order. All food items not in the established taxonomic categories, plus any terrestrial invertebrates (i.e., winged adults presumably taken from the surface) were classified in the category "other." Approximately two thirds of all items categorized as "other" were actually adult terrestrial invertebrates.

Table 1—Numbers of stomachs examined per fish per species and number of individuals of that species surveyed at 12 sites on four creeks in the Ouachita National Forest (1989-1991)[a]

Species	Stomachs	Number observed
Etheostoma radiosum Orangebelly darter	270	1982
Lythrurus umbratilis Redfin shiner	226	1,764
Semotilus atromaculatus Creek chub	185	2,798
Luxilus chrysocephalus Striped shiner	137	1,183
Notropis boops Bigeye shiner	95	704
Fundulus olivaceus Blackspotted topminnow	88	229
Etheostoma blennioides Greenside darter	79	170
Etheostoma collettei Creole darter	72	212
Etheostoma whipplei Redfin darter	67	209
Notropis ortenburgeri Kiamichi shiner	64	321
Lepomis megalotis Longear sunfish	55	425
Labidesthes sicculus Brook silverside	54	57
Fundulus catenatus Northern studfish	31	41
Noturus lachneri Ouachita madtom	31	37
Lepomis cyanellus Green sunfish	25	52
Esox americanus Grass pickerel	24	30
Aphredoderus sayanus Pirate perch	19	19
Hypentelium nigricans Northern hog sucker	14	40
Erimyzon oblongus Creek chubsucker	12	176
Micropterus dolomieui Smallmouth bass	11	44
Ameiurus natalis Yellow bullhead	10	13

[a] All seining and snorkeling observations pooled.

Our analyses are based on 1,589 fish stomachs from field samples in 1989-91. Fish stomachs were distributed among study streams as follows: Saline Drainage, South Fork Alum (non-harvested), 425 stomachs and Bread Creek (harvested), 304 stomachs; Little Missouri Drainage, Crooked Creek (non-harvested), 331 stomachs and Blaylock Creek (harvested), 529 stomachs. Estimates of abundances of the 15 most common species, excluding the central stoneroller, were derived from seining and from supplementary snorkeling observations made at all sites by Gelwick in spring 1991. These abundance estimates were used to extrapolate estimates of effects. Snorkeling observations also were used to derive classifications of species as "benthic" or "surface and water column" species. The central stoneroller, a common species in study streams, was excluded from our analyses because it depends almost exclusively on an algal diet.

Data analyses included calculations of a standard diversity measure (Shannon-Weiner) and a percent similarity index to compare and contrast foods eaten by assemblages and species among streams and between harvest regimes. Determination of overall similarities and differences among species was examined by multivariate clustering and principal components analyses using the NT-SYS computer programs. Our study was aimed at first approximations of food use by fishes in the study streams and required an exploratory approach to data analysis. As such, data analysis is summarized with results so that the method and rationale can be addressed in the context of principal findings.

RESULTS

Stomach contents were represented by 10 to 270 individuals each of 21 common fish species (table 1). The majority of invertebrate taxa recovered from fish stomachs were dipteran (midge) larvae and pupae, which occur on stream substrates, or adult insects that primarily are available to fishes at the water's surface (> 6,000 items represented by dipterans) (fig. 1). Other commonly eaten food items included (in order of abundance in stomachs): Ephemeroptera (mayflies); Trichoptera (caddisflies); Homoptera (true bugs); Arachnoidea (spiders, ticks, and mites); Crustacea (mostly crayfish and excluding microcrustaceans); Coleoptera (beetles); Hymenoptera (ants, wasps, and bees); Lepidoptera (moths and butterflies); microcrustaceans (mostly cladocerans and copeopods); Annelida (segmented worms); Plecoptera (stoneflies); and Nematoda (nematodes). Eight other invertebrate categories occurred frequently enough to be included in the analyses (fig. 1).

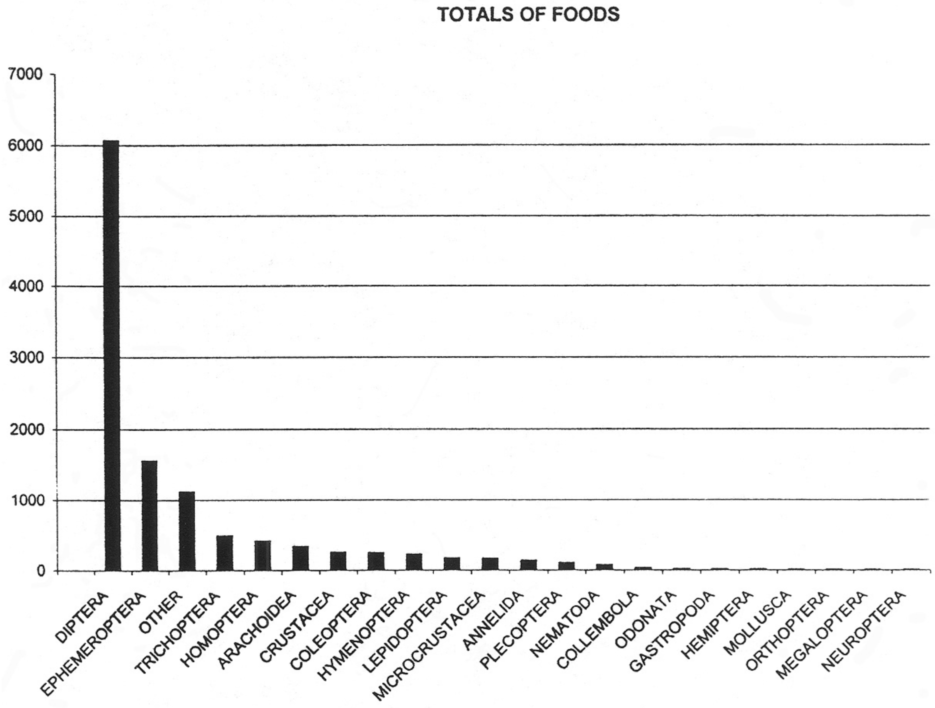

Figure 1—Total number of food items by taxonomic category found in stomachs of 1,589 fishes examined in four streams of the Ouachita National Forest.

A principal components analysis (PCA) was carried out for the 21 most common fish species to determine similarities and differences among species in foods eaten across 18 taxonomic categories of food items (with finer invertebrate categories pooled within orders). In PCA, each food item may be visualized as an individual axis across species that is aggregated with related food items into a lesser number of new or principal component axes (PC Axes), each of which represents original items that were most highly correlated with the new axis (Gauch 1982). The PCA provides a view of similarities and differences in diet by fishes, and in two dimensions, arrays fish species in multivariate space along gradients of foods used by those fishes. The first three PC axes accounted for 49.2 percent of the total variance in food items among the 21 fish species (19.1, 15.6, and 14.5 percent, for PC Axes I, II, and III, respectively). Accounting for nearly half the total variance in food use is a relatively high result for an ecological analysis. To compare fish species with respect to their overall similarities and differences on these "food" axes, two-dimensional scatterplots of fish species scores on PC Axis I versus PC Axis II and PC Axis I versus PC Axis III were used to interpret food use by fishes (figs. 2A and 2B).

PC Axis I separated (fig. 2A, from right to left) fish species with a tendency to eat Homoptera, Annelida, Hymenoptera, or "other" items (two thirds of which were terrestrial) from species that fed less on these categories. PC Axis II (fig. 2A) separated species that fed more on Ephemeroptera, Trichoptera, Lepidoptera, and Nematoda from species that ate less from these groups and more crustaceans. Interpretation of the distribution of fish species along the food axes is facilitated if benthic (bottom living) species are delineated from surface and water column species (fig. 2C). Surface and water column species are almost completely separable in overall diet from benthic species (fig. 2C). Surface and water-column species included: *Luxilus chrysocephalus* Rafinesque; *Lythrurus umbratilis* (Girard); *Fundulus olivaceus* (Storer); *Notropis boops* Gilbert; *Semotilus atromaculatus* (Mitchill); *Labidesthes sicculus* (Cope); *Fundulus catenatus* (Storer); and various sunfish and bass species. Benthic species included: darters such as *Etheostoma blennioides* Rafinesque, *Etheostoma radiosum* (Hubbs and Black), *Etheostoma whipplei* (Girard), and *Etheostoma collettei* Birdsong and Knapp; catfishes like *Noturus lachneri* Taylor and *Amieurus natalis* (Lesuer); and the sucker, *Hypentelium nigricans* (Lesueur). The only exception to this separation in diet is the sucker *Erimyzon oblongus* (Mitchill), which is a mostly benthic species but was grouped on both food use axes with the surface and water column species. Overall, the PCA axes suggest that the benthic species eat more aquatic insect larvae (e.g., mayflies, caddisflies, or moth larvae), whereas the surface and water column species feed more on items of terrestrial origin. Although not surprising, this finding underscores the high diversity in food use by a local fish assemblage. Importantly, the patterns emphasize that food use, which is indicative of potential role in the ecosystem, relates in a predictable fashion to the microhabitat use by each species.

PC Axis III accounted for almost as much variance as did PC Axis II. Thus, it was of interest to examine distribution of fish species on those two axes (figs. 2B and 2D). The two axes did, in fact, provide greater overall separation of species in multivariate diet space with less evidence of outliers. Importantly, use of some of the most common food items,

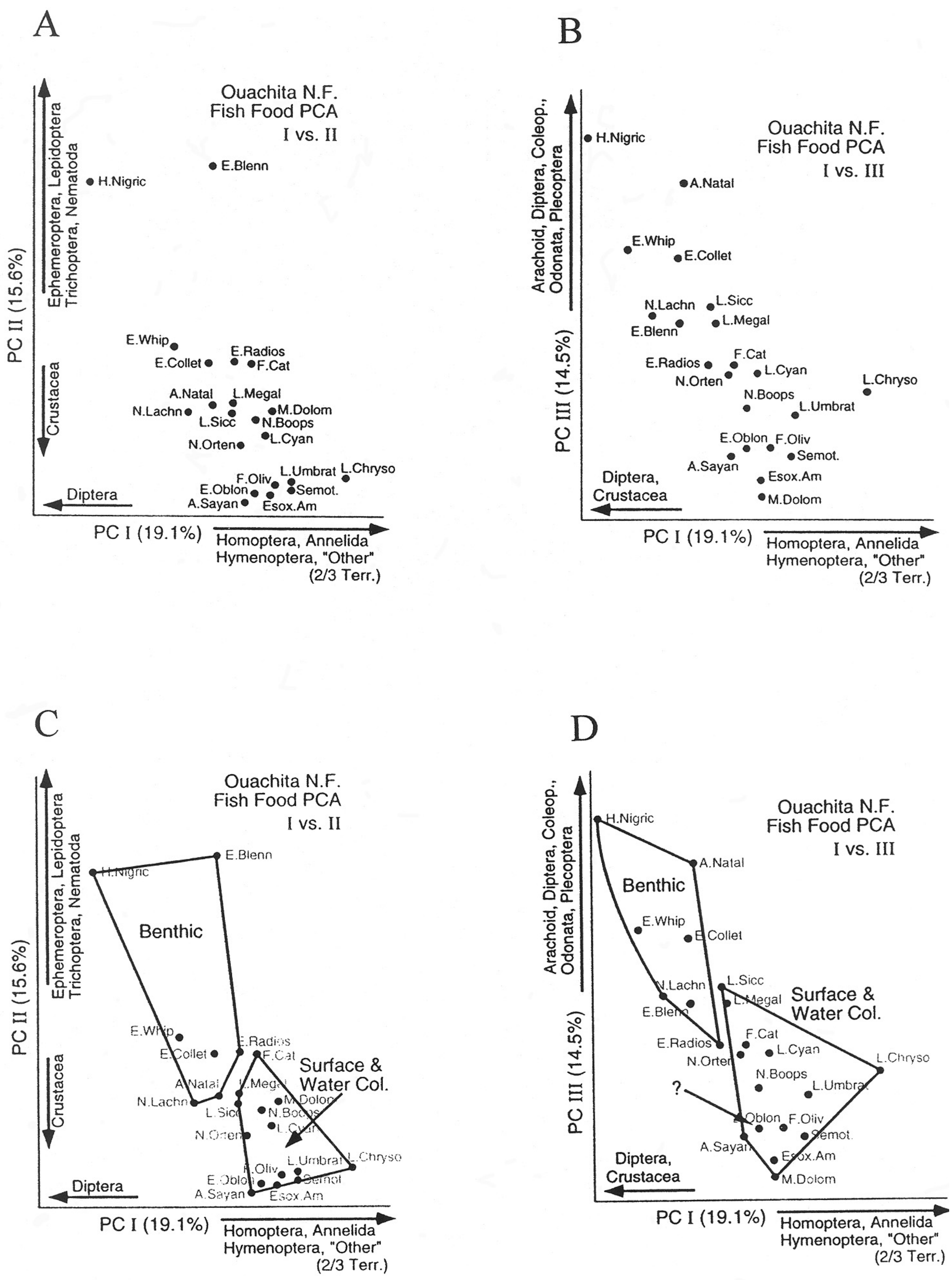

Figure 2—Principal components analysis ordinating fish species on the basis of food use in four streams of the Ouachita National Forest (A). PC Axis I verus PC Axis II; (B). PC Axis I versus PC Axis III; (C). PC Axis I verus PC Axis II, but with benthic and water column species denoted by polygons; (D). PC Axis I versus PC Axis III, but with benthic and water column species denoted by polygons.

including dipterans, was highly associated with PC Axis III. PC Axis III separated species along a gradient of high to low consumption of dipterans, odonates, plecopterans, or coleopterans (upper to lower, respectively, fig. 2B). Again, benthic species (e.g., *Hypentelium nigricans, Amieurus natalis, Etheostoma whipplei, Etheostoma collettei*) fed more on these items than did the surface and water column species. Combination of PC axes II and III provided complete segregation of benthic from surface and water column species on the basis of diet (fig. 2D). Again, *Erimyzon oblongus* was an exception (fig. 2C). It grouped with the surface and water column species in diet in spite of its benthic microhabitat use. The PCA results indicate distinctive separation in diet of most species that coincides with microhabitat use. From this, it is suggested that managers may be able to make rough approximations of trophic roles of fish in stream ecosystems by using the location of fish species in the water column.

Principal component analysis also indicated some correspondence between general food consumption and taxonomic association of species. From the delineation of the three most abundant families (darters, Percidae; minnows, Cyprinidae; sunfishes, Centrarchidae) on the PCA plots (figs. 3A and 3B), it is clear that darters are strongly segregated in diets from the sunfish and minnows. Obviously, this can relate to the differences in microhabitat use. Darters live and feed almost exclusively on the stream bottom predominantly in riffle habitats, and minnows and sunfishes live and feed more in the water column of pools. However, it is equally apparent that there is substantial overlap between minnows and sunfishes in overall diet. Their overlap in diet is logical in that these two families include species that overlap strongly in occurrence in the water column (on the basis of our snorkeling observations), and it is obvious in the field that members of both families will take food items (e.g., adult insects) from the surface of a pool. It would be tempting to conclude that managers could merely sample a stream, determine the family to which a species belonged, or the degree of relatedness between species, and assume some information about the similarities and differences in roles of the species in the ecosystem. This appears not to be the case.

Two triangular similarity matrices were created to more rigorously test the hypothesis that food use among fish species is predictable from taxonomic relatedness. One matrix measured similarity on the basis of overall diet, and the other, taxonomic distances among the species. Similarity in diets for the 21 fish species was assessed by a pair-wise product moment correlation matrix of foods eaten by food category. The diet similarity matrix (data not shown) was summarized by clustering of fish species (fig. 4). Taxonomic distances among species were scored for all possible species pairs using criteria from Douglas and Matthews (1992). Species within the same genus were scored with a taxonomic distance of 1; species in the same family, as 2; species in same order, as 3; and species from different orders, as 4. The resulting taxonomic similarity matrix gave each species pair a score of 1 to 4, depending on the relatedness of the species. If similarities (or differences) in diet among species pairs are associated with taxonomic relatedness, then the two matrices (diet and taxonomic similarity) should match closely. To test the degree of association between the matrices, a Mantel test was performed (Douglas and Matthews 1992) in which one matrix was randomly rearranged 9,999 times, and the degree to which the real matrices match each other was compared to the way the matrices would

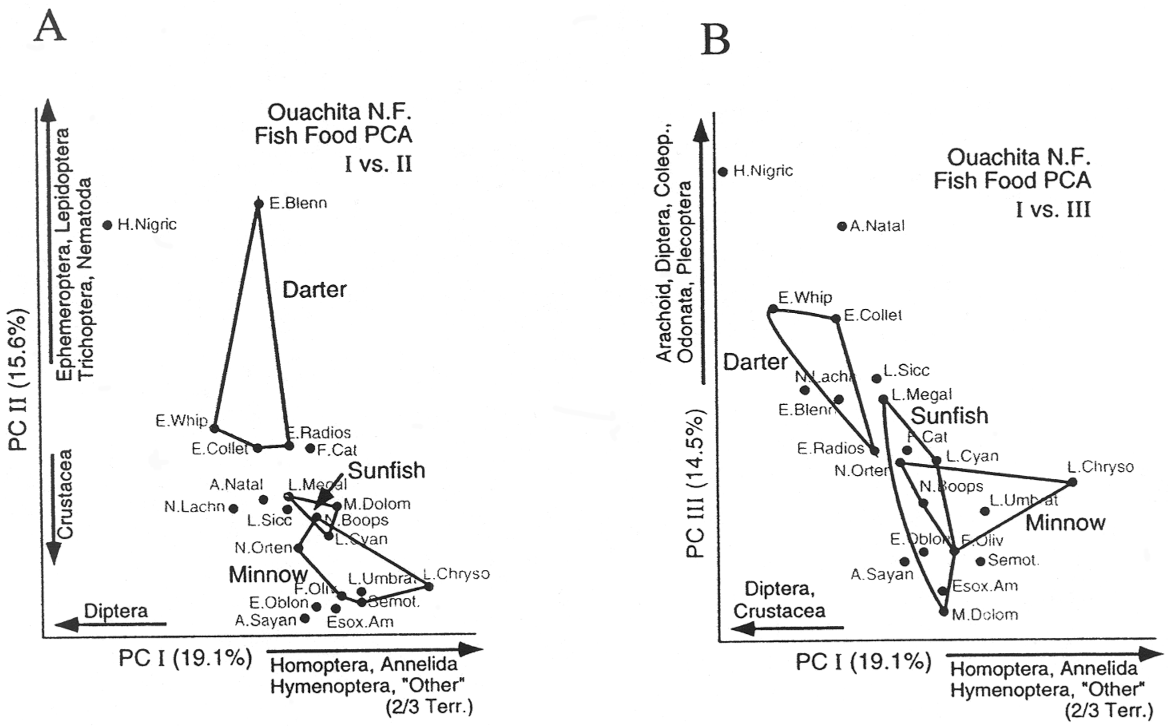

Figure 3—Principal components analysis as in figure 2 but with most abundant families denoted by polygons.

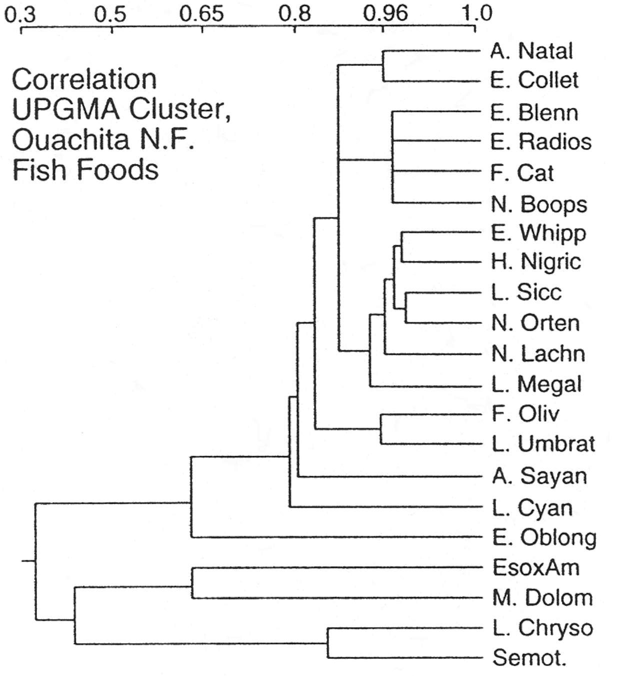

Figure 4—Phenogram based on correlation of food use similarities between 21 common species in four streams of the Ouachita National Forest.

match after being “reshuffled” at random. The Mantel test of the food use and taxonomic similarity matrices was not significant ($P = 0.395$) which signifies the two matrices are not closely or significantly matched in structure. Although there are some general differences in food use by darters versus minnows and sunfishes (explained earlier), there was no statistically detectable correspondence between diet and taxonomic relatedness across all species. Hence, managers could not safely predict trophic similarities among species on the basis of taxonomy alone. Each common species in an ecosystem needs detailed study of diets or their function in that respective system.

The degree of redundancy in a stream fish assemblage or among species within a region is another question of interest. Redundancy implies that closely related species might be so similar in their trophic or functional ecology that one could be substituted for the other. For example, if two closely related species are fully redundant, it would be possible for one to be lost from a system, the other to be introduced, and the system retain all of its original functional properties. In this study, the redundancy hypothesis can be assessed by examining positions of closely related fish species in multivariate food use space (figs. 2A and 2B). The PCA plots indicate some pairs of closely related species are rather similar in diets relative to other species in general (fig. 2B) including the minnows, *Notropis boops* and *Notropis ortenburgeri* Hubbs; the sunfishes, *Lepomis megalotis* (Rafinesque) and *Lepomis cyanellus* Rafinesque; and the minnows, *Luxilus chrysocephalus* and *Lythrurus umbratilis.* Conversely, other taxonomically similar species pairs are broadly separated in multivariate diet space (fig. 2B) including: the darters, *Etheostoma radiosum* and *Etheostoma whipplei*; the suckers, *Hypentelium nigricans* and *Erimyzon oblongus*; the catfishes, *Noturus lachneri* and *Amieurus natalis*; and the topminnows, *Fundulus catenatus* and *Fundulus olivaceus*. Overall, there is no apparent support for the postulate that closely related species pairs

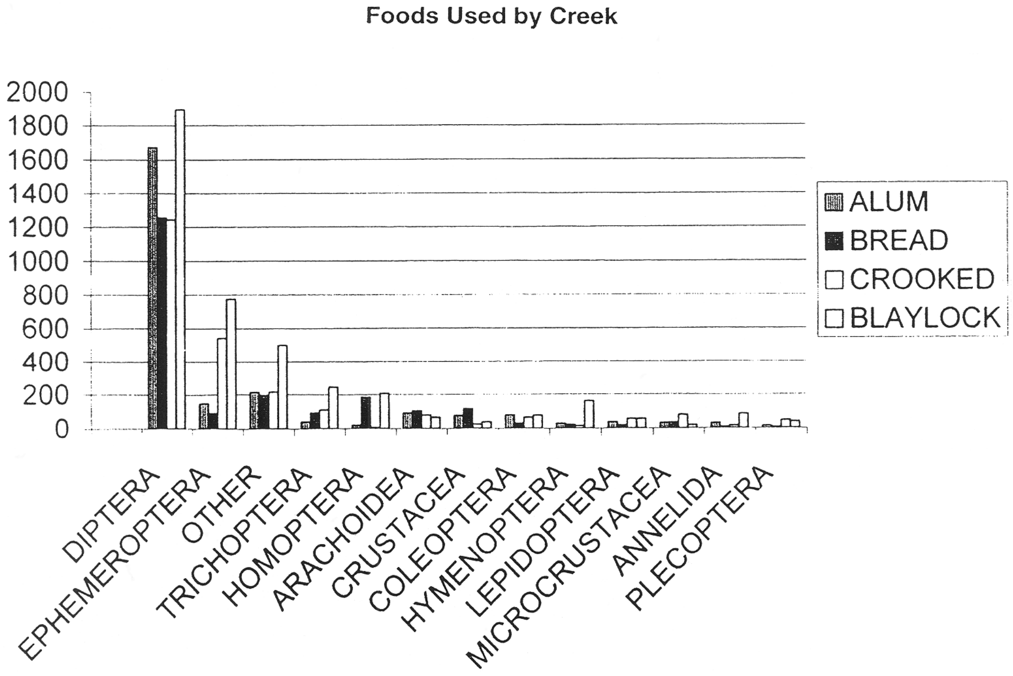

Figure 5—Use of 13 major food categories by fish assemblages across four streams of the Ouachita National Forest. Vertical (Y) axis represents total numbers of each item found an all examined fish stomachs.

are highly similar in diets. They do not in general appear redundant or "substitutable" for each other. For management, this implies intentional or accidental introductions of species are not likely to allow one species to be lost and another gained without potential for trenchant changes in ecosystem function.

Because context-specific diet of fishes influences their potential ecosystem effects, the degree of similarity of trophic effects for fish assemblages in different creeks is another important management consideration. To address this, our entire sample of fish stomach contents was summarized separately in each creek across all four sampling periods (fig. 5). In all four creeks, dipterans dominated in diets of the 21 species (fig. 5), and the "other" category (mostly adult insects) also was important in all creeks. Differences in importance of mayflies (Ephemeroptera) in fish diets appeared to be related to river drainage. Far fewer mayflies were eaten by fish in Alum and Bread Creeks (Saline River drainage) than in Crooked and Blaylock Creeks (Little Missouri drainage). However, mayfly consumption may be related more to typical environmental conditions in the streams than to specific drainage. Alum and Bread Creeks have seasonally intermittent flows (Taylor and others, in press), (Williams and others, in press), whereas Crooked and Blaylock Creeks always exhibited substantial flows during our visits. Persistent flow of the latter streams should favor more consistent availability of mayflies, particularly in riffles (Brown and Brussack 1990) which may dry up periodically in Alum and Bread Creeks. This hypothesis is further supported by the lower occurrence of Trichoptera (caddisflies; also linked to flowing habitats) in diets of fishes in Alum and Bread Creeks than Crooked and Blaylock Creeks (fig. 5). Besides simple taxonomic composition, diversity of diets can reflect breadth of potential impact of a fish assemblage in a stream. However, Shannon-Weiner diversity indices of fish stomach contents (pooled by creek) were not substantially different among creeks, ranging from 1.42 in Alum Creek to 1.81 in Blaylock Creek. Thus, some foods of fishes differed among streams, likely in the context of the environmental setting which influences food availability, but the dominant food item (dipterans) was the same in all four streams.

To assess the degree to which timber harvest or drainage influences overall similarity in fish diets, a Percent Similarity Index (PSI) was calculated for all food items across stomachs in each creek. Within a river drainage, total fish assemblages in harvested and non-harvested creeks showed high similarity in diets. In the Saline drainage, similarity between foods eaten in Alum (non-harvested) and Bread (harvested) Creeks was 85.1 percent. Likewise, the PSI for Crooked (non-harvested) and Blaylock (harvested) Creeks was 71.5 percent. Creeks with similar harvest history, but in different river drainages, also showed high similarity values (harvested, Bread and Blaylock, PSI = 74.5 percent; non-harvested, Alum and Crooked, PSI = 77.0 percent). Thus, there was no evidence that timber harvest histories as defined here influenced similarities or differences in foods eaten by fish assemblages among creeks.

Intraspecific difference in diet is the finest scale of resolution to examine context-specific effects of fishes on stream ecosystems. For selected species, diet was examined to determine if intraspecific differences could be related to timber harvest regime or creeks. For each of the seven most abundant species in our samples, similarity (PSI) was calculated between stomach contents for all individuals in harvested and non-harvested watersheds (within the river drainage where the species was present and/or most abundant, since most species did not occur in all creeks or both drainages). Intraspecific PSI in diet between harvested and non-harvested watersheds was high; five of seven species had PSI values of > 5 percent. Values ranged from a high of 87.8 percent for *Etheostoma radiosum* to as low as 48.0 percent for *Lepomis megalotis*. PSI values for other species were: *Etheostoma whipplei*, 84.7 percent; *Etheostoma collettei*, 81.9 percent; *Semotilus atromaculatus*, 78.5 percent; *Fundulus olivaceus,* 76.2 percent; and *Lythrurus umbratilis,* 63.1 percent. Intraspecific diversity (Shannon-Weiner Index) of diets for these species also was similar in harvested and non-harvested watersheds. There is no compelling evidence from our data set that timber harvest regimes, at the spatiotemporal scale evaluated here, had strong effects on foods consumed by individual fish species. Similarity between streams was highest for the three darter species (*Etheostoma* spp.), which feed primarily on benthic organisms in riffles, and lowest for three species (*Fundulus olivaceus, Lythrurus umbratilis*, and *Lepomis megalotis*) that may feed substantially on surface items (e.g., adult terrestrial insects). It could be speculated that disturbance from timber harvest changes availability of terrestrial insects more than that of riffle dwelling benthic invertebrates, but direct supporting evidence or plausible mechanisms are lacking.

For a single species, patterns were sought in food use within a stream, longitudinally, seasonally, and in regard to harvest regime. The largest sample of stomachs was available for *Etheostoma radiosum*, which is abundant throughout Crooked and Blaylock Creeks. A PCA with raw data standardized within food item was conducted for this species where each sample at a site (Fall 1989; May 1991, when most specimens were available) was ordinated along gradients of the eleven most used food items. PC Axes I and II accounted for 25.7 and 18.0 percent of the variance, respectively, but the results suggested few local patterns in food use. Visual inspection of plots of PC Axis I versus PC Axis II (not shown), revealed no difference in amount of multivariate diet space occupied by *E. radiosum* in either creek but did indicate a slightly greater diet breadth in May than in October. No strong longitudinal patterns in diet were detected. Likewise, diets in extreme headwaters (where few or no other species are present) were not notably different from those at larger, downstream sites.

Fish density and known foods were used to calculate a crude estimate of the number of invertebrates eaten by a local fish assemblage. The estimate is instructive in evaluating direct and indirect impacts fish food consumption may have in these stream ecosystems. In May 1991, Gelwick counted fish by snorkeling while seine sampling was conducted at the 24 field sites. The total counts by seining and snorkeling for the 15 most common species (excluding *Campostoma*) are provided by Matthews (1993). Potential use of invertebrate foods by fish in a 100-m reach of the study streams were estimated for each species as: (individuals/m) x (detection factor) x (food items/stomach) x (feedings/d).

Individuals per meter is the total numbers of a species detected by snorkel or seine (using greatest value) divided by 4800 m of stream sampled (approximately 200 m per site at all 24 sites). A detection factor of four was used to account for undetected individuals. Detection of 25 percent of individuals actually present at a sampling site is reasonable in our judgement, but it is emphasized that an increase or decrease in this factor will drastically change the final estimate. Food items per stomach were based on the average number of items found in stomachs for a species, and feedings per day were estimated as six. Based on food passage times of minnows (Unpublished data. On file with: William Matthews, Professor, University of Oklahoma Biological Station, HC 71 Box 205, Kingston, OK 73439), the observed food items in each stomach were assumed to represent foods ingested in the last two hours before the fish were sampled and for purposes here, represents one feeding. Our use of six feedings per day is derived from studies that show most minnows and darters feed throughout daylight hours (Matthews and others 1982; Surat and others 1982). The resulting product was summed across species, and the sum expressed as invertebrate foods used per 100 m of stream.

Our calculations yielded a crude estimate of 13,144 invertebrate food items consumed by a “typical” fish assemblage (i.e., 15 common species) per 100 m of stream per day. Clearly, this does not reflect reality at any one site but is a very general extrapolation from stomach contents and sites that only broadly represent streams in the Ouachita National Forest. Importantly, even a minor adjustment in assumptions of detection or number of feedings can drastically affect the estimate. However, if 13,000 items per l00 m of stream per day is viewed as an “order of magnitude” estimate, a starting point is established for exploring the degree to which fishes remove invertebrates from these stream systems or the rates of transfer of nutrients from living invertebrates to release in the water column. Exploration of these and similar dynamic rates could be useful in modeling overall trophic effects of fishes in stream ecosystems of the region.

DISCUSSION

Many studies have shown the trophic plasticity of stream fishes, but most also revealed strong relationships between diet and microhabitat use of stream fishes. Mendelson (1975) found, for example, that minnows differed in habitat selection, but fed opportunistically on invertebrates available in those microhabitats. Our findings support the hypothesis of correspondence between microhabitat use by stream fishes and their potential trophic effects within the ecosystem. Importantly, our study also suggests that food use by a species is not fully predictable from taxonomic relationships. We summarize other major findings as follows:

1. Dipterans and terrestrial insects were major food items eaten by fish assemblages in all creeks
2. For 18 major invertebrate food items, individual species were distinctive in feeding habits
3. Major families of fishes were generally distinctive in diets.
4. Similarity in foods eaten was not rigidly related to taxonomic relatedness of species, and closely related species within families are not redundant or substitutable in stream ecosystems.
5. For seven common fish species, diets were 75 percent similar on average between watersheds with little or no timber harvest and those with timber harvest conducted under Forest Service standards and guides in the last 20 yr.
6. Likewise, overall diets were 70 to 85 percent similar between fish assemblages in harvested and non-harvested watersheds.
7. At least one common, widespread species (orangebelly darter) showed little longitudinal or spatial pattern in diets use within a stream.
8. Average food intake by each species, multiplied by estimated densities of the species over all study streams yielded a crude estimate of 13,000 food items eaten per day per 100 m of stream.

Our study produced evidence related to trophic effects of fish in ecosystems. Expanded field and manipulative studies are beginning that ask more detailed questions about functional effects observed from snorkeling observation of feeding, microhabitat use, and behaviors of fishes in these ecosystems. Our goal in the next several years is to expand what is now known about food habits and context to focused experiments linking fish species to stream processes from a functional groups perspective. This experimental approach will eventually incorporate in stream experiments and observations, focused mechanistic experiments, and tests of ecosystem roles of fishes in artificial stream systems.

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THE ROLE OF REGIONAL FACTORS IN STRUCTURING OUACHITA MOUNTAIN STREAM ASSEMBLAGES

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Abstract—We used Basin Area Stream Survey data from the USDA Forest Service, Ouachita National Forest to evaluate the relationship between regional fish and macroinvertebrate assemblages and environmental variability (both natural and anthropogenic). Data were collected for three years (1990-1992) from six hydrologically variable stream systems in the Ouachita Mountains that were paired by management regime within three drainage basins. Most of the variability in regional fish assemblages was explained by the historically constrained drainage basins themselves rather than measured habitat variables. Macroinvertebrate assemblages also showed some historical constraint but were associated more closely with stream habitat conditions. Timber harvest regimes showed little effect on regional assemblages. At the basin-level, taxonomic assemblages of both fishes and macroinvertebrates were a better predictor of environmental variability than trophic assemblages, bringing into question the use of trophic groups as an ecological measure in large-scale studies. Regional analyses are critical for understanding how stream systems are organized at different spatial scales and are important for effective management of streams within an altered landscape.

INTRODUCTION

Describing the interactions that determine distribution and abundance of organisms is a primary goal of ecologists (Andrewartha and Birch 1954, Townsend and others 2000) and has defined much of the field of stream ecology (Allan 1995, Gorman and Karr 1978, Tonn and others 1983). These relationships, however, are complex and vary depending on spatial, temporal, or taxonomic resolution (Schlosser 1987, Vinson and Hawkins 1998, Wiens and others 1986). The assembly of regional stream faunas differs from that of local assemblages (Tonn 1990). At larger space and time scales, biota are influenced more by historical events (e.g., biogeographical history and speciation events) and climatic factors (Cooper and others 1998, Hugueny 1997, Lohr and Fausch 1997, Ricklefs and others 1999, Vinson and Hawkins 1998). These large-scale, historical influences act as “filters” for regional faunas and limit the total species pool available at smaller scales (Cueto and de Casenave 1999, Ricklefs and others 1999, Tonn 1990).

At smaller scales, local ecological processes (i.e., abiotic and biotic) act as a final “filter” in structuring assemblages (Tonn 1990). The role of both environmental factors (Gorman and Karr 1978, Grimm and Fisher 1989, Grossman and others 1982, Jackson and Harvey 1993) and biotic interactions (Dahl and Greenberg 1998, Gilliam and others 1989, Power and Matthews 1983) in structuring aquatic assemblages is well documented for stream systems. Because of the dependence on factors operating at larger space and time scales, it is overly simplistic to only consider the role of local processes in organizing assemblages. Knowledge of regional and historical influences are critical for refining hypotheses of community assembly and organization for streams at smaller scales (Angermeier and Winston 1998, Brooks and McLennan 1993, Caley and Schluter 1997).

We used a large data set from the USDA Forest Service to examine patterns of regional fish and macroinvertebrate assemblage structure. The data were collected from six hydrologically variable streams paired within three drainage basins in the Ouachita National Forest, AR. Because much of the flow in these streams is dependent on rainfall events, they are prone to flood in the spring and dry to isolated pools in the summer. Variable flow regimes can strongly influence the structure of stream assemblages (Boulton and others 1992, Capone and Kushlan 1991, Delucchi 1988, Stanley and others 1994, Taylor 1997).

In addition to the natural disturbance regime, streams in the Ouachita National Forest are impacted by anthropogenic disturbances, primarily activities associated with timber harvesting. The effects of timber harvesting practices on stream fishes in the central United States are poorly understood (Brown and Matthews 1995, Hicks and others 1991, Rutherford and others 1992). Macroinvertebrates inhabiting forested streams are particularly responsive to timber harvesting (Wallace and others 1997, Webster and others 1992), and responses may include short-term shifts in dominant functional groups and/or changes in community structure (Allan 1995, Campbell and Doeg 1989). Timber harvesting practices can have both short and long-term effects on stream biota, with most long-term effects resulting from persistent sedimentation (Campbell and Doeg 1989, Silsbee and Larson 1983).

In this study, we used multivariate techniques to examine the relationship between regional fish and macroinvertebrate assemblages and environmental variability (both natural and anthropogenic) in these small, Ouachita Mountain stream systems. We were particularly interested in how the assemblages would differ in their response to

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this variability. Fishes are constrained to their historically defined drainage basins, and we predicted that this historical constraint would strongly influence their response to environmental variability. In contrast, many macroinvertebrates emerge from the aquatic environment as adults and choose future breeding sites based on environmental quality at a stream locality (Anderson and Wallace 1984). Macroinvertebrates have the ability to rapidly recolonize a site following a disturbance event (e.g., flood or drought; Stanley and others 1994). Because of their recolonization potential and the ability to cross basin barriers, macroinvertebrate assemblages are less constrained by large-scale processes, and their response to environmental variability should differ from that of fishes. Previous authors (Lammert and Allan 1999, Plafkin and others 1989) have suggested that macroinvertebrates should be more strongly associated with local habitat conditions and fishes influenced more by factors operating at larger spatial scales.

Finally, we were interested in how these patterns change with assemblage resolution (taxonomic versus trophic groupings). Because assemblages based on trophic groups would be less constrained by phylogeny and biogeographic history, we predicted that they may be more strongly associated with variability in habitat conditions than taxonomic assemblages.

MATERIALS AND METHODS

Study Area

The six streams in this study are located within the Saline, Cossatot, and Arkansas River basins in the Ouachita Mountains, Ouachita National Forest, AR (table 1). The Ouachita Mountains, located in southeastern Oklahoma and southwestern Arkansas, are a series of east-west oriented ridges and valleys. The mountains are composed of Paleozoic sedimentary rock, and streams are dominated by bedrock, boulder, and cobble substrata with some finer sediments interspersed (Robison 1986). Many small streams in the Ouachita Mountains have a highly variable flow regime throughout the year (Brown and Matthews 1995, Taylor 1997). As most of these streams are maintained primarily by rainfall events, they are prone to flooding in the spring and drying to isolated pools in the summer months.

Within each of the three drainage basins, streams were paired with one in a basin managed for timber and the other in a basin with no harvesting activities (Ponce and others 1982), (table 1). The extent of timber harvest activities over the last 100 years is reflected in age-class distributions of trees within the basins (table 1). Relative to reference basins, managed basins show higher areal percentages of forest in early succession (i.e., trees less than 50 years old) and a lower percentage of forest in mid to late-successional stages (Clingenpeel 1994). Because a variety of harvesting practices are used, managed basins in the Ouachita National Forest are a mosaic of stand-types ranging from small clearcuts to late-successional stands. Reference basins are also a mosaic of stand ages, but natural processes dictate the structure of these stands.

Sampling Methods

All six study streams were sampled each summer (late May to early August) from 1990-1992 by Forest Service inventory teams (led by JAC). A mesohabitat classification system (McCain and others 1990) was used to divide each stream into habitat units (e.g., mid-channel pool, backwater pool, run, glide, high gradient riffle, cascade) from their headwaters to downstream reaches.

Physical stream features were measured within each classified habitat unit. Stream width was measured along a transect at the midpoint of each habitat unit, and depth was measured at four equidistant points and the thalweg along this same transect. Several instream variables were visually estimated as the percentage of area occupied within a habitat unit, including: substrate composition, embeddedness, undercut banks, large and small woody debris, whitewater, boulder ledges, terrestrial vegetation overhanging the stream, vegetation clinging to substrate, rooted vegetation, and bank stability (i.e., percent of the bank not eroded). Bank angle and canopy closure were determined at the center of each habitat unit. Clingenpeel (1994) provided a detailed description of these methods.

Water samples were collected in 10 percent of each type of habitat unit (e.g., if 300 mid-channel pools were present, 30 were sampled). These collections were stratified along the length of each stream (Clingenpeel and Cochran 1992).

Table 1—Streams sampled by the USDA Forest Service in 1990-1992

Stream	Basin	Area	Mgmt.	Early	Late successional
		ha		- - - - - - *percent*	- - - - - -
South Alum	Saline	1533	R	9	55
Bread	Saline	1517	M	40	20
Caney	Cossatot	2518	R	2	70
Brushy	Cossatot	3428	M	32	18
Dry	Arkansas	2170	R	8	30
Jacks	Arkansas	2938	M	25	20

Reference streams indicated by ‘R’ and managed by ‘M’, with this assignment based on the percentages of each basin in early (less than 50 years old) versus late successional (greater than 80 years old) forest.

Dissolved oxygen and temperature were measured in the field. Water samples were analyzed for conductivity, pH, bromide, nitrate, phosphorus, manganese, magnesium, sodium, cobalt, calcium, and sulfate at Berea, KY, with 1983 EPA Methods for Chemical Analysis of Water and Wastes.

Fishes and macroinvertebrates also were collected in 10 percent of each habitat unit type, arrayed longitudinally in each stream. Fishes were collected with multiple-pass electrofishing and block nets (Clingenpeel and Cochran 1992) and preserved for identification. Macroinvertebrates were collected at each locality with a kick-net and substrate washing into a net (5-min each). These samples were pooled and preserved for later identification.

Statistical Methods

We summarized stream habitat and water chemistry variables for each stream by year and computed means for each. An overall mean and coefficient of variation were calculated for depth measurements (transect and thalweg). We used 39 physical variables in a principal components analysis (PCA) to reduce the dimensionality of these data to a few significant axes. We were not attempting to interpret these axes; rather, we used PCA to objectively select a small subset of the original data that would represent the major gradients in the study streams. We retained six variables (conductivity, percent canopy cover, percent boulder substrate, CV depth, percent cover of rooted vegetation, and bank stability) for further analyses. These variables had the highest component loadings for the first five axes, which accounted for about 72 percent of the total variance. We retained conductivity and canopy cover because they both had equally high loadings on the first axis.

We also summarized fish and macroinvertebrate data for each stream by year. For taxonomic analyses, we used fish species and macroinvertebrate genera (or order or family when genus was not determined). We classified fishes and macroinvertebrates into trophic groups according to Horwitz (1978) and Merritt and Cummins (1984), as summarized by Allan (1995).

We used Canonical Correspondence Analysis (CCA), (ter Braak 1986, 1990) to examine the relationship between regional fish and macroinvertebrate assemblages (taxonomic and trophic) and the reduced set of environmental variables. We also included dummy environmental variables for the individual river basins (i.e., Arkansas, Cossatot, Saline) and presence or absence of timber harvesting within a watershed. These analyses allowed us to simultaneously examine effects of the historically defined drainage basins, presence or absence of logging activities, and measured environmental variables on assemblage structure. Based on assemblage structure and the environmental variables modeled, we plotted the site scores for each stream by year in multivariate space and grouped them by drainage basin. We constructed these graphs separately for fish and macroinvertebrate trophic and taxonomic assemblages. We used Monte Carlo tests (1000 permutations) to estimate the significance of each CCA performed.

We used the Mantel test (Fortin and Gurevitch 1993) to examine the association between assemblage structure (taxonomic and trophic) and measured environmental variables. We were interested specifically in how strongly the trophic matrices were associated with both taxonomic and habitat matrices. We tested the prediction that trophic matrices would be associated more strongly with environmental variability than taxonomic matrices because of their relative freedom from phylogenetic and historical constraint. For the Mantel test, we used rectangular n x n matrices (e.g., fish taxa by site) to construct triangular correlation matrices. The Mantel test determines the association between pairs of the triangular correlation matrices at a time, testing the null hypothesis that there is no association within elements of the matrices. Thus, for these data we tested the association between trophic assemblages and both taxonomic assemblages and environmental variability (i.e., trophic vs. taxonomic and trophic vs. habitat). We also performed partial Mantel tests (Manly 1997), where a given test between two matrices is constrained by removing association with a third matrix (e.g., trophic vs. habitat with the effects of taxonomy removed). Partial Mantel tests are comparable in function and interpretation to partial correlation analyses (Manly 1997, Zar 1996). We repeated these Mantel tests for both fish and macroinvertebrate assemblages.

RESULTS AND DISCUSSION

Summary of Collections

A total of 30 species of fishes (9 families) and 152 genera of macroinvertebrates (65 families, 20 orders) were collected over the 3-year study period from the six streams. Examples of all major trophic groups (Horwitz 1978, Merritt and Cummins 1984, Allan 1995) were represented in these data. For both fishes and macroinvertebrates, overall number of individuals collected was highest in streams of the Cossatot River basin (table 2). For fishes, species richness was higher in the managed streams than in their paired reference for all three basins (table 2). In two of three cases (Saline and Arkansas basins), richness of macroinvertebrate taxa was greater in the managed streams than the paired reference (table 2). However, because of high variability among basins, there were no significant differences (T-test, $p > 0.05$) between managed and reference watersheds with respect to number of individuals collected or richness of fishes or macroinvertebrates.

Fish Assemblage Structure

Based on taxonomic and trophic assemblages of fishes, streams within individual drainage basins were closely associated and separated from other basins in multivariate space (fig. 1). Thus, most of the variation in regional fish assemblages was explained by differences among the historically defined drainage basins rather than measured environmental variables or timber harvesting activities. For both groups of assemblages, variation in the first CCA axis was dominated by the individual drainage basins (table 3), indicating that assemblages differed strongly among the historically defined basins. The second CCA axis was associated with the percentage of boulder substrate and the drainage basins (table 3). Logging regime was correlated weakly with both axes, indicating little effect in structuring fish assemblages at the regional scale.

Table 2—Total number of individuals collected and taxa richness for each of the six study streams over the 3-year study period

Measure	Stream/Basin					
	S. Alum Saline	Bread* Saline	Caney Cossatot	Brushy* Cossatot	Dry Arkansas	Jacks* Arkansas
Fishes						
No. Indiv.	1,694	849*	6,320	7,020*	921	1,550*
Richness	18	19*	16	19*	4	13*
Invertebrates						
No. Indiv.	2,402	4,821*	11,648	10,962*	2,185	3,611*
Richness	38	47*	43	40*	28	44*

* = managed streams.

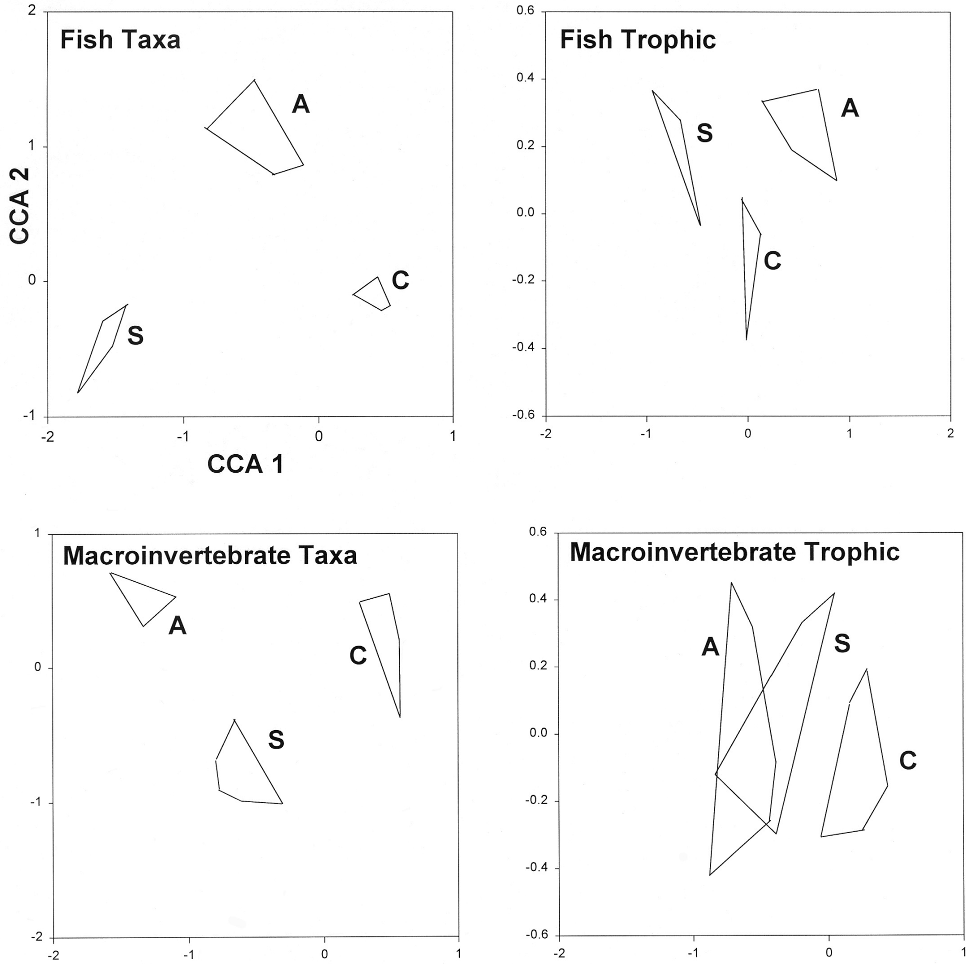

Figure 1—The distribution of site scores in multivariate space for fish and macroinvertebrate assemblages (taxonomic and trophic). Streams within a given year are grouped with polygons by drainage basin. A = Arkansas basin; C = Cossatot basin; S = Saline basin. Shown are the first (x-axis) and second (y-axis) canonical correspondence analysis axes.

Overall, streams within individual basins differed in environmental conditions. For example, streams in the Saline basin had greater aerial canopy cover and more rooted vegetation. Streams in the Arkansas basin were characterized by a higher percentage of boulder substrate and greater bank stability, and streams in the Cossatot system had higher conductivity and greater variability in stream depth (L.R. Williams and others. Environmental variablility, historical

Table 3—Correlations between canonical correspondence axes (1 and 2) and environmental variables for fish and macroinvertebrate assemblages (taxonomic and trophic)

Variable	Fish taxa		Fish trophic		Macroinvertebrate taxa		Macroinvertebrate trophic	
Axis	1	2	1	2	1	2	1	2
P-value	(0.01)	(0.01)	(0.01)	(0.02)	(0.01)	(0.02)	(0.01)	(0.01)
Conductivity	0.42	0.03	0.10	-0.16	0.43	-0.17	0.35	0.08
Boulder	-0.33	-0.68	0.42	0.63	-0.78	0.42	-0.81	-0.28
Canopy	-0.56	-0.30	-0.51	0.40	-0.29	-0.25	-0.19	-0.04
CV depth	0.55	0.09	0.40	-0.01	0.60	0.09	0.60	0.42
Vegetation (rooted)	-0.43	-0.37	-0.62	0.13	0.07	-0.87	-0.06	0.48
Stability (bank)	-0.28	0.34	0.22	0.27	-0.53	0.35	-0.43	-0.10
Basin								
Alum	-0.89	-0.42	-0.80	0.33	-0.41	-0.74	-0.38	0.41
Cossatot	0.91	-0.38	0.15	-0.71	0.94	0.28	0.85	-0.26
Logging	0.03	0.17	-0.03	0.11	-0.05	-0.41	-0.15	0.27

contingency, and the structure of regional fish and macroinvertebrate assemblages in Ouachita Mountain stream systems. Manuscript in preparation). Based on the relationships we found, regional fish assemblages are likely influenced by combinations of spatially constrained environmental variability (i.e., habitat conditions unique to individual basins) and the historical biogeography of individual drainage basins.

Macroinvertebrate Assemblage Structure

Taxonomic assemblages of macroinvertebrates showed a similar pattern to fish assemblages, but the separation of streams into their respective basins was not as evident for macroinvertebrate trophic groups (fig. 1). For macroinvertebrate taxa, the first CCA axis was again dominated by differences among the historically defined drainage basins and the percentage of boulder substrate (table 3). The second axis was correlated with drainage basins and percent cover of rooted vegetation (table 3). Macroinvertebrate trophic groups were also associated with drainage basins; the Cossatot basin clearly separated from the other two in multivariate space (fig. 1). Percentage of boulder substrate and variability in stream depth also had a strong influence on trophic assemblages (table 3). Logging regime was not associated strongly with regional macroinvertebrate assemblages.

The weak relationship between macroinvertebrate trophic assemblages and drainage basins, and stronger associations with individual environmental variables, indicates that historical constraint may be less important in structuring regional macroinvertebrate assemblages. In contrast with fishes, macroinvertebrate assemblages were associated more strongly with environmental variability not associated with the drainage basins (i.e., habitat conditions that are unrelated to basins). Despite these stronger correlations with individual environmental variables, historical contingency still had some influence on macroinvertebrate assemblages when grouped by taxa (table 3).

Trophic Patterns

For fish assemblages particularly, patterns for taxonomic and trophic assemblages were quite similar (fig. 1). This was in contrast to our prediction that trophic assemblages would be less constrained by historical contingency than taxa. Mantel tests indicated that fish and macroinvertebrate trophic groups were associated with both environmental variables and the taxonomic composition of assemblages (table 4). Because the trophic matrices are constructed from species matrices, we focused on the strengths of the relationships (r) and not associated p-values. As expected, the strongest correlations were between trophic and taxonomic assemblages (table 4). Correlations between trophic groups and habitat were not as strong, and when the effect of taxonomy was removed, there was little correlation left between trophic assemblages and environmental variables (table 4). Thus, correlations between trophic and habitat matrices were confounded by their strong relationships with the taxonomic matrices. This indicates that taxonomic assemblages were associated more strongly with environmental variability than the trophic assemblages, and this pattern was evident for both fishes and macroinvertebrates.

Table 4—Correlations between trophic assemblages for fishes and macroinvertebrates and taxonomic and habitat matrices

Matrix comparison	r
Fish trophic groups x taxa	0.701
Fish trophic groups x habitat	0.343
Partial Mantel (taxonomy effects removed)	0.012
Macroinvertebrate trophic groups x taxa	0.659
Macroinvertebrate trophic groups x habitat	0.212
Partial Mantel (taxonomy effects removed)	0.133

Correlations derived from Mantel and partial Mantel tests.

CONCLUSIONS

Overall, we found that timber harvesting regime had little influence on number of individuals collected, taxonomic richness, or assemblage structure of fishes and macroinvertebrates at the regional scale. The lack of an effect seems to indicate that these systems are somewhat resistant to such perturbations, and we believe this is related to the dynamic natural disturbance regime of these streams. At the large scale of this study, the seasonal flooding and drying of these streams appears to override effects of timber harvesting activities as practiced in the Ouachita National Forest. However, land-use activities often have a stronger effect on stream biota at local rather than regional scales (Campbell and Doeg 1989, Lammert and Allan 1999, Rutherford and others 1992). Because of limitations in sampling design, we were unable to test this prediction at smaller spatial scales.

As we predicted for fishes, historical constraint had the strongest influence on the structure of regional assemblages. This historical explanation, however, has two components that are difficult to separate. First, biogeographical history influences the potential biota of a stream (Brooks and McLennan 1993, Matthews and Robison 1988, Tonn 1990). Second, habitat conditions within individual streams are constrained by the underlying geology of individual basins (Brown and Matthews 1995, Brussock and others 1985); basin geomorphology determines substrate composition and channel characteristics. Both of these historical components (i.e., biogeography and habitat variability unique to individual basins) are important in structuring regional fish assemblages in the six streams that we examined.

In contrast, regional macroinvertebrate assemblages were structured both by historical factors and also environmental variability that was not associated with the individual basins. We did not expect such a high degree of historical constraint on macroinvertebrate taxa. As above, this may relate to the inherent differences in habitat characteristics of the individual basins.

Regional studies of stream systems are critical for understanding the structure of their biotic assemblages. Huston (1999) argued that unraveling the interrelationships between local and regional processes is critical to understanding global patterns of species diversity. How these ecological processes are linked at different spatial scales is a fundamental question in the field of ecology (Levin 1992, Lohr and Fausch 1997). Because of the inherent difficulty in managing small stream systems (Poff and Ward 1989), detailed studies at multiple spatial and temporal scales are necessary to conserve and manage their biological diversity. An understanding of the effects of disturbance is also necessary for effectively managing aquatic ecosystems (Resh and others 1988). One of the basic tenets of ecosystem management is that disturbance is a vital attribute of ecological systems and native species have evolved within the context of this natural disturbance regime (Hessburg and others 1999, Landres and others 1999). The effects of anthropogenic disturbances cannot be fully understood unless the natural disturbance regime has been taken into consideration (Landres and others 1999). Because many government agencies are using basin-level survey methods (Clingenpeel 1994, Hankin and Reeves 1988) to assess effects of anthropogenic disturbances, it is important to evaluate their efficacy. Given the intensive sampling methodologies and expense of these basin-level studies, it is important to consider the strengths and limitations of these methods in elucidating ecological relationships.

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FISH ASSEMBLAGE STRUCTURE UNDER VARIABLE ENVIRONMENTAL CONDITIONS IN THE OUACHITA MOUNTAINS

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Abstract—Spatial and temporal variability of fish assemblages in Ouachita Mountain streams, Arkansas, were examined for association with stream size and flow variability. Fishes and habitat were sampled quarterly for four years at 12 sites (144 samples) in the Ouachita Mountains Ecosystem Management Research Project, Phase III watersheds. Detrended and Canonical Correspondence analyses were used to describe spatial and temporal patterns. Considerable spatial and temporal variability in fish assemblage structure was observed among sites. Across all sites and samples, assemblage variability was significantly associated with stream size and flow regime. The flow regime of the Little Glazypeau system differed from that of the Alum Fork system. Differences in flow regime were significantly associated with differences in fish assemblages across sites with trenchant differences noted between the Little Glazypeau and Alum Fork systems. The two systems are historically distinct and reflect large-scale differences in geomorphology, speciation, extinction, and dispersal.

INTRODUCTION

The Ouachita Mountains of Arkansas and Oklahoma contain a suite of river systems that ultimately drain into the Red and Arkansas River basins. The region supports a diverse fish fauna with several endemic species (Robison and Buchanan 1988). The implementation of the Ouachita Mountains Ecosystem Management Research Project (Phase III) by the USDA Forest Service provided us with the opportunity to study how environmental variability influences fish assemblage structure across a large spatial scale incorporating the four Phase III core watersheds.

Many factors affect the distribution and abundance of species, including interaction between the environment and population processes (Brown 1984). Determining environmental influences on distribution and abundance patterns is a difficult task because different spatial and temporal scales yield different types of information (Wiens and others 1986). Regardless, differences among species in distribution and abundance patterns within a region are, in part, due to individualistic responses to the environmental template. In streams, two complex environmental gradients appear to influence fish assemblages and habitat structure: stream size and hydrologic variability (Gorman and Karr 1978; Horwitz 1978; Poff and Ward 1989; Schlosser 1987; Sheldon 1968). Small, hydrologically dynamic streams in these Ouachita Mountain watersheds provide an ideal system to address questions concerning the effects of environmental variability on the distribution and abundance of fishes.

Our primary objective for this work was to examine how local fish assemblages change across the landscape and to see if these patterns were related to spatial and temporal variability in the environment. Specifically, we used multivariate direct and indirect gradient analyses to ask if patterns in fish assemblage structure were associated with stream size and variability in flow.

MATERIALS AND METHODS

Study Area and Species

The Ouachita Mountains of Arkansas and Oklahoma are characterized by strongly folded, uplifted sedimentary rock and pine-oak upland forest (Robison 1986). Our sample sites were located in two river systems (of the Red River drainage) that drain the eastern side of the uplift. Nine sites were located in the Alum Fork of the Saline River (hereafter Alum Fork) and three sites were in the Little Glazypeau system of the Ouachita River drainage (fig. 1). These are

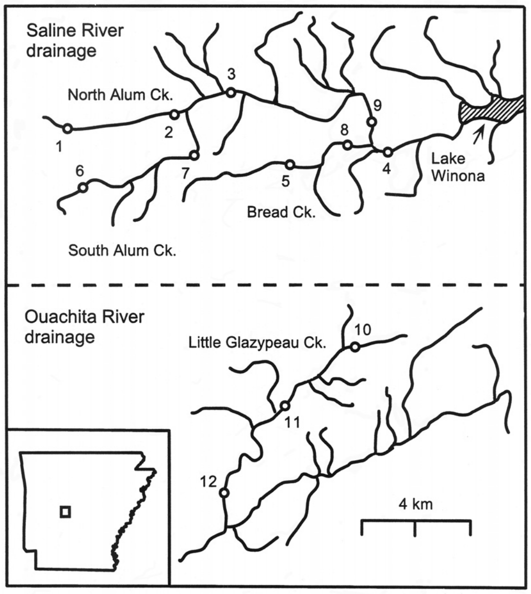

Figure 1—Map showing 12 sites in the Alum Fork and Little Glazypeau systems. Dashed line indicates that the geographic proximity of the two systems is not as shown.

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clear water streams except during high precipitation events when dramatic increases in turbidity are evident. Because of the high gradient of these streams, riffle-pool development is strong. Pools ranged from a few to 50 m or more in length, and were usually separated by swift riffle habitats consisting of coarse substrate materials. These streams are characterized by high variability in the flow regime. Water levels can rise and fall very rapidly with heavy precipitation events and headwater reaches are intermittent, frequently drying to isolated pools during summer and autumn months. Thus, habitats and their corridors can grow, shrink, and change configuration rapidly (Taylor 1997).

Data Collection

From November 1995 through August 1998, we sampled fishes and measured environmental variables at the 12 sites (fig. 1) for a total of 144 samples. Samples for all years were taken in November, February, May, and August and were always completed within a three to four day time period under ambient conditions. At each site, we electrofished all habitat types within a 100 to 200-m stream reach (depending on stream size) that spanned at least two pool-riffle sequences. For stream fishes, the stream reach is a logical sample unit in a river system (Frissell and others 1986), and the effects of disturbance on assemblage level properties are measured most commonly at this scale (Matthews 1998). All sampling methods are biased and electrofishing is no exception. However, our system consisted of small, clear streams (first to third order) that were wadeable, and that could be sampled in a consistent manner each time. The same individual (CMT) always operated the backpack electroshocker and there were always two to three netters present to pick up stunned fishes. Thus, we are confident that we adequately assessed the composition of fish assemblages at a given site for each sample date. After sampling a site, all fishes were identified, counted, and returned to the stream alive.

After fish sampling and processing, we measured habitat variables along transects (perpendicular to stream flow) spaced at 10 to 15-m intervals along the sampled stream reach. The number of transects varied (eight to 12) depending on the length of the sampled stream reach. Transects were permanently marked and re-surveyed during each visit. At every other meter along transects, we measured current velocity, depth, and dominant substrate type. We categorized substrate following Taylor and Lienesch (1996a, 1996b) as bedrock, large boulder (> 300 mm), small boulder (150 to 300 mm), cobble (50 to 149 mm), gravel (3 to 49 mm), or sand (< 3 mm). We measured stream width at each transect as well as the presence/absence of aquatic macrophytes, algae, undercuts, and woody debris.

Data Treatment and Analyses

In order to summarize the fish assemblage data and quantify the effects of stream size and flow variability on fish assemblages, we performed three multivariate analyses. Detrended correspondence analysis (DCA) is an indirect ordination technique designed to summarize complex community data (Gauch 1982). We used DCA to qualitatively compare fish assemblages across space and time. Canonical correspondence analysis (CCA) is a multivariate direct gradient technique that is commonly used to explore the relationship between assemblages of organisms and the environment. The ordination axes (gradients in assemblage structure) are constrained by the suite of environmental variables being analyzed, thus maximizing the species-environment correlation (ter Braak 1986). All analyses were performed with PC-ORD software (McCune and Meford 1997).

The first CCA was designed to quantify the association between stream size and fish assemblage structure. To quantify stream size we first calculated the mean and maximum width, mean and maximum depth, and mean and maximum current velocity for each site/date. All environmental values were log-transformed before analysis. Our goal was to see if patterns in fish assemblage structure were associated with stream size variables. We used a Monte Carlo randomization procedure to determine if the relationship was significant (1000 permutations).

Next, we used CCA to identify patterns in assemblage structure that reflect variation through time in stream width, depth, and flow characteristics. We included coefficients of variation (CV) calculated for each site across sample dates for the following variables: mean and maximum width, mean and maximum depth, and mean and maximum current velocity. The species data used in this analysis included summed totals for each species at each site across the entire study period. We used a Monte Carlo randomization procedure to determine if the relationship between assemblage structure and flow variability was significant (1000 permutations). Multiresponse permutation procedures (McCune and Meford 1997) were used to determine if there were assemblage-level differences between the Little Glazypeau system and the Alum Fork system with regard to the flow regime.

RESULTS AND DISCUSSION

We collected a total of 30 fish species (and numerous individuals of a hybrid sunfish, *Lepomis sp.*) from the 12 sites. Species were taxonomically distributed across six families and functionally distributed across six trophic groups (Allan 1995). We present species relative abundances and distributional information in table 1 and summary environmental data in table 2.

Species varied greatly in their overall relative abundances and in their distribution across sites (table 1). At the coarsest level, five species were collected only from the Little Glazypeau system, and six species were collected only in the Alum Fork system. Large-scale historic differences in geomorphology, speciation, extinction, and dispersal account for the non-shared components of the assemblages. Both systems drain into the Red River, but are isolated by long distances of lowland, big river habitat. The effectiveness of isolation is demonstrated by the evolution of one endemic form (*Noturus lachneri*, Alum Fork system) and several species that occur in one system but not the other. Within a given drainage abundances and occurrences were also highly variable across space and time, reflecting the different ecological conditions that occur spatially and temporally. These patterns are exemplified in the following indirect and direct gradient analyses.

Table 1—Species, species acronyms, species occurrences in the two drainage systems, total number of localities occupied (occurrence) by species, and overall relative abundances of species

Species	Acronym	Little Glazypeau	Alum Fork	Occurrence	Relative abundance
Ameiurus natalis (Lesueur)	AMENAT	X	X	7	< 0.01
Aphredoderus sayanus (Gilliams)	APHSAY	X	X	9	0.01
Campostoma anomalum (Rafinesque)	CAMANO	X	X	11	0.18
Chaenobryttus gulosus (Cuvier)	LEPGUL		X	1	< 0.01
Erimyzon oblongus (Mitchill)	ERIOBL	X	X	12	0.04
Esox americanus Gmelin	ESOAME	X	X	10	0.01
Etheostoma blennioides Rafinesque	ETHBLE	X	X	3	< 0.01
Etheostoma collettei Birdsong & Knapp	ETHCOL		X	6	0.07
Etheostoma radiosum (Hubbs & Black)	ETHRAD	X		3	0.05
Etheostoma whipplei (Girard)	ETHWHI		X	9	0.09
Fundulus catenatus (Storer)	FUNCAT	X		2	0.01
Fundulus olivaceous (Storer)	FUNOLI	X	X	10	0.05
Hypentelium nigricans (Lesuer)	HYPNIG	X	X	3	< 0.01
Ichthyomyzon gagei Hubbs and Trautman	ICHGAG		X	3	< 0.01
Labidesthes sicculus (Cope)	LABSIC	X	X	6	0.01
Lepomis cyanellus Rafinesque	LEPCYA	X	X	12	0.07
Lepomis hybrid	LEPHYB	X	X	5	< 0.01
Lepomis macrochirus Rafinesque	LEPMAC	X	X	8	0.01
Lepomis megalotis (Rafinesque)	LEPMEG	X	X	11	0.19
Lythrurus umbratilis (Girard)	LYTUMB	X	X	10	0.07
Luxilus chrysocephalus Rafinesque	LUXCHR	X		3	0.01
Micropterus dolomieu Lacepede	MICDOL	X		1	< 0.01
Micropterus punctulatus (Rafinesque)	MICPUN	X	X	6	< 0.01
Micropterus salmoides (Lacepede)	MICSAL	X	X	5	< 0.01
Notropis boops Gibert	NOTBOO	X	X	7	0.02
Notropis ortenburgeri Hubbs	NOTORT		X	3	0.01
Noturus nocturnus Jordan & Gilbert	NOTNOC	X		1	< 0.01
Noturus lachneri Taylor	NOTLAC		X	7	0.05
Percina caprodes (Rafinesque)	PERCAP	X	X	5	0.01
Pimephales notatus (Rafinesque)	PIMNOT	X	X	5	0.02
Semotilus atromaculatus Mitchill	SEMATR	X	X	9	0.04

Table 2—Substrate characteristics (percent composition of point estimates), mean stream width, mean stream depth, and mean stream current speed for the 12 sites

	Substrate								
Sites	Sand	Gravel	Cobble	Small boulder	Large boulder	Bed-rock	Mean width	Mean depth	Mean current speed
1	0	30	25	15	10	20	2.6	16.7	0.07
2	0	0	43	36	14	7	3.7	21.9	0.15
3	8	11	70	11	0	0	4.9	34.0	0.14
4	0	18	33	24	6	18	10.8	39.4	0.15
5	0	3	53	29	15	0	4.0	29.1	0.13
6	0	10	23	45	23	0	3.7	30.0	0.02
7	0	16	16	42	26	0	6.7	40.0	0.09
8	0	3	21	21	9	47	6.0	30.0	0.07
9	0	0	21	38	41	0	7.9	36.1	0.19
10	0	62	31	4	0	4	2.8	22.1	0.05
11	3	26	43	23	3	3	4.9	28.6	0.13
12	0	34	6	23	37	0	11.3	34.7	0.13

Substrate compostion is reported for November 1996 samples under ambient flow conditions. Means for other variables are calculated across the entire study period.

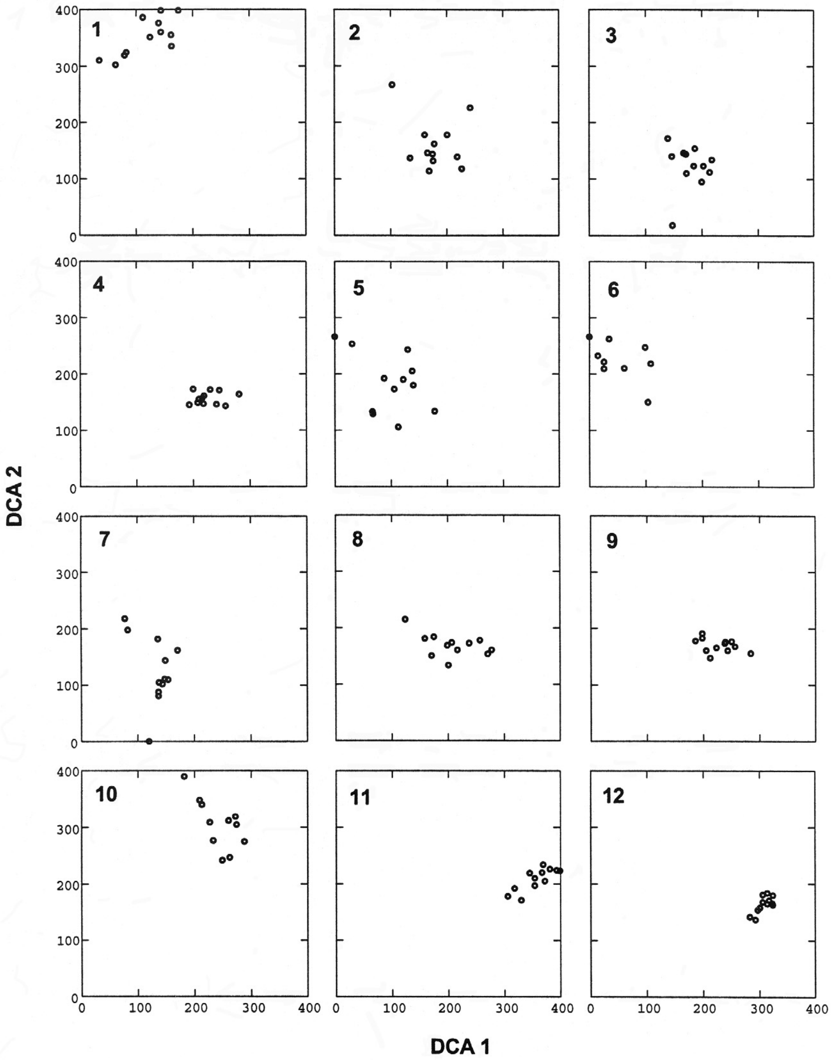

Figure 2—Detrended correspondence analysis (DCA) of all site and samples across the three year period. Sample scores are plotted in multivariate space defined by DCA axes 1 and 2.

Detrended correspondence analysis of all samples at each site indicated considerable variability in the structure of site-specific assemblages across time (fig. 2). The overall large spread of points in the two-dimensional space was indicative of high species turnover (beta diversity) among the sites. This was especially evident when comparing sites from the Little Glazypeau system (sites 10, 11, and 12) with sites from the Alum Fork system (sites one through nine). For samples from individual sites, the variability was considerably less at larger sites as compared to smaller sites. Sites 4, 9, and 12 were the largest sites (see mean width, table 2) and samples from these sites clustered together very closely in ordination space (fig. 2). In contrast, samples from the smallest sites (e.g. sites 1, 2, 5, 6, and 10) were much more spread out in ordination space. This pattern indicated that assemblages at smaller sites were much more variable across time in species composition and abundance than assemblages from larger sites. Despite variability in assemblage structure for each site across time, each locality was confined to a relatively small proportion of the overall two-dimensional ordination plot (fig. 2).

Interpretation of the variability in assemblage structure is facilitated by simultaneously examining the ordination plot (fig. 3) and correlations of species abundance with the two DCA axes (table 3). For example, *Semotilus atromaculatus* (SEMATR) has the highest species score on DCA axis 2 (top of figure 3). Samples from Site 1 (fig. 2) all clustered in this region of the ordination space indicating the importance of this species at this site. As expected, *S. atromaculatus* showed a strong positive correlation with DCA axis 2 (r = 0.576; table 3). *Lepomis cyanellus* (LEPCYA) also was an important species at site 1 and showed a strong positive correlation with DCA axis 2 (r = 0.399). *Etheostoma radiosum* (ETHRAD) had the highest score on DCA axis 1 (far right in figure 3) and was found only in the two largest sites in the Little Glazypeau system (sites 11 and 12, figure 2). This species showed a very strong correlation with DCA axis 1 (r = 0.620). By superimposing sites in figure 2 onto figure 3 and examining the correlations in table 3, a visual picture emerges of the species that are important assemblage components at the 12 localities.

Table 3—Pearson correlations of species abundance and first two DCA ordination axes

Species	Axis 1 correlations	Axis 2 correlations
Ameiurus natalis	0.378	-0.152
Aphredoderus sayanus	-0.253	-0.270
Campostoma anomalum	0.582	-0.349
Chaenobryttus gulosus	-0.058	-0.042
Erimyzon oblongus	-0.290	-0.307
Esox americanus	-0.363	-0.126
Etheostoma blennioides	0.166	-0.117
Etheostoma collettei	-0.027	-0.326
Etheostoma radiosum	0.620	0.103
Etheostoma whipplei	-0.531	0.128
Fundulus catenatus	0.508	-0.058
Fundulus olivaceous	0.137	-0.423
Hypentelium nigricans	0.454	-0.042
Ichthyomyzon gagei	-0.008	-0.137
Labidesthes sicculus	0.039	-0.138
Lepomis cyanellus	0.120	0.399
Lepomis hybrid	-0.186	-0.030
Lepomis macrochirus	0.040	-0.015
Lepomis megalotis	0.359	-0.364
Lythrurus umbratilis	-0.085	-0.473
Luxilus chrysocephalus	0.564	0.016
Micropterus dolomieu	0.347	-0.115
Micropterus punctulatus	0.240	-0.165
Micropterus salmoides	0.081	-0.086
Notropis boops	0.012	-0.220
Notropis ortenburgeri	-0.155	-0.365
Noturus nocturnus	0.154	-0.029
Noturus lachneri	-0.076	-0.367
Percina caprodes	0.204	-0.127
Pimephales notatus	0.138	-0.212
Semotilus atromaculatus	-0.143	0.576

DCA = Detrended correspondence analysis.

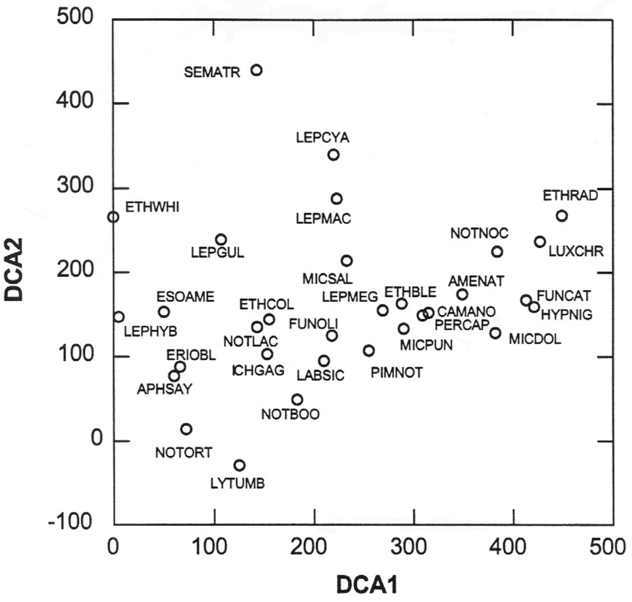

Figure 3—Detrended correspondence analysis (DCA) of all site and samples across the three year period. Species scores are plotted in multivariate space defined by DCA axes 1 and 2.

CCA allowed us to examine the importance of stream size and hydrologic variability in structuring fish assemblages. Correlations between the environmental variables and CCA axes one and two (table 4) indicate the relative contribution of each variable to the CCA axes. For the stream size analysis, the Monte Carlo randomization test indicated that the relationship between the stream size variables and assemblage structure was highly significant for the first two axes (p = 0.001, both axes). Thus, assemblage structure was strongly associated with stream size.

CCA also indicated that variation in stream flow was associated with assemblage structure. A Monte Carlo randomization procedure indicated that the relationship between flow variability and assemblage structure was significant for the first CCA axis (p = 0.054) and marginally significant for the

Table 4—Environmental correlates of first two CCA axes from two separate analyses[a b]

	Size		Variability	
Variable	CCA Axis 1	CCA Axis 2	CCA Axis 1	CCA Axis 2
Mean width	0.440	0.756	-0.339	0.157
Maximum width	0.238	0.925	0.382	0.041
Mean depth	0.054	0.614	0.466	-0.432
Maximum depth	0.502	0.441	0.594	-0.564
Mean current	0.193	0.326	0.041	0.558
Maximum current	0.076	0.462	0.833	0.408

CCA = Canonical correspondence analysis.

[a] For the size analysis all environmental variables represent means, calculated for each site across all sample dates.

[b] For the variability analysis all environmental variables represent coefficients of variation, calculated for each site across all sample dates.

second CCA axis (p = 0.104). The strongest correlate was variability in maximum current velocity (table 4), and this vector separated the largest sites (4, 8, 9, 11, and 12) from the other sites (larger sites showed less variability in maximum current velocity). Other strong correlates on the first CCA axes were related to CVs of depth, width, and current (table 4). Thus, variability in the flow regime (CVs of stream size and current speed) was also an important predictor of local fish assemblage structure.

An interesting pattern emerged between drainage systems when sites were plotted in two-dimensional space defined by CCA axes 1 and 2 from the stream flow analysis. Assemblages from the three sites in the Little Glazypeau system (sites 10, 11, and 12) separated from the rest of the sites based on variables associated with the flow regime (fig. 4).

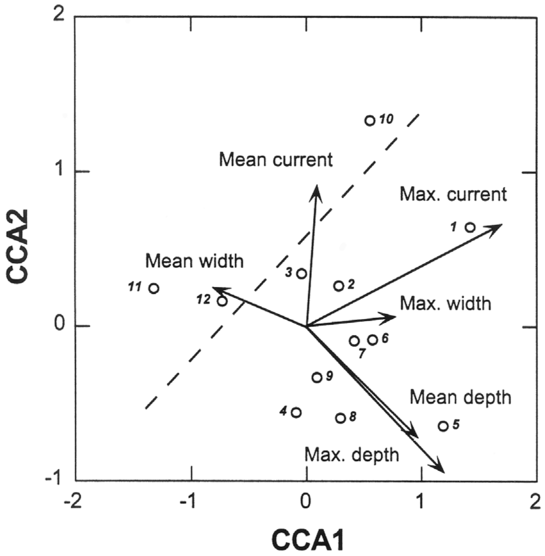

Figure 4—Canonical correspondence analysis (CCA) of 12 sites. Environmental variables are coefficients of variation, calculated across the 3-year period. Species abundances are summed across time for each site.

Multi-response permutation procedures indicated that this separation was significant (p = 0.032). Thus, the Little Glazypeau system differed from the Alum Fork system in its variability with respect to the flow regime, and these differences were also associated with differences in fish assemblage structure.

CONCLUSIONS

Our results suggested that assemblages in these Ouachita Mountain streams were strongly associated with factors related to stream size and flow variability. Furthermore, Taylor and Warren (Taylor, C.M.; Warren, M.L. Dynamics in species composition for stream fish assemblages: environmental variability and nested species subsets. Manuscript in preparation) found that stream size and flow variability were correlated negatively for these streams. Thus, hydrologic variability was spatially structured in this system. Smaller stream localities were more variable with respect to flow than larger stream localities, and this variation was associated with the structure of the resident fish assemblages. Further, the Alum Fork and Little Glazypeau systems showed significant differences in assemblage structure that were associated with variability in the flow regime. The Little Glazypeau system generally has less variable flows than the Alum Fork system due to geological differences in groundwater discharge. The two systems also differed for historical reasons. Little Glazypeau and Alum Fork systems are geographically separated, and this separation has produced differences in the regional species pools.

ACKNOWLEDGMENTS

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CONSEQUENCES OF POOL HABITAT ISOLATION ON STREAM FISHES

David G. Lonzarich, Melvin L. Warren, Jr., and Mary E. Lonzarich[1]

Abstract—For fishes, stream habitat units (i.e., pools and riffles) often exist as relatively discrete patches of varying quality that are distributed in a mosaic along the stream continuum. Under these conditions, the possibility exists that the spacing of suitable patches within a stream reach may affect interhabitat movements of fishes and their pattern of distribution within habitat patches. We summarized the findings of two different, but related, studies that demonstrated how isolation of pools by long and shallow riffles affected recolonization and daily movement patterns of pool dwelling fishes in two Arkansas streams. Our results show for the first time that the spatial distribution of habitats within streams can significantly affect the abundance of species and characteristics of fish assemblages within pool habitats. Because land use activities can alter habitat spacing, our findings also have important implications for fish conservation in degraded streams.

INTRODUCTION

Characteristics of fish assemblages within stream habitats are shaped by physical and biological factors that operate at different temporal and spatial scales (e.g., Poff 1997). Stream ecologists are obligated to determine habitat variables that most influence fish and the spatial scale(s) over which they operate (Rabeni and Sowa 1996). To make predictions, stream researchers have long used information about the physical characteristics of streams and knowledge of the hierarchical nature of the stream environment (e.g., Frissell and others 1986, Naiman and others 1992). Physical features of habitat units (e.g., pools, riffles) such as area or volume (Schlosser and Angermeier 1989), depth, water current, cover, and substrate (Gorman and Karr 1978), often correlate strongly with patterns of fish diversity, richness, and biomass. At large spatial scales (e.g., reaches, watershed), important factors may include flow regime (e.g., Horwitz 1978), temperature, valley form, distance to mouth, and zoogeographic history.

Species distribution and assemblage patterns are shaped by these natural controls and changes in the physical environment that can accompany land use activities. Land use practices, especially those associated with the removal of riparian vegetation, can lead to changes in stream temperature, flow, nutrient input, and habitat quality (Hicks and others 1991, Schlosser 1991). In degraded streams, reduced availability of habitat forming woody debris can reduce the abundance of pools in a stream (e.g., Keller and Swanson 1979, Schlosser 1991). A potentially important, though poorly appreciated, consequence of such a change is that the loss of pool habitat also reduces the frequency (Ralph and others 1994) and therefore increases the spacing (e.g., Beechie and Sibley 1997) of these habitats within the stream.

Although the ecological impacts of land use disturbances have been widely studied, the historical focus was on small scale impacts occurring within stream habitats or reaches (e.g., Schlosser 1982, Hicks and others 1991). Recent papers have addressed some of the effects of land use activities on habitat connectivity and fragmentation of streams (e.g., Schlosser 1991, Rieman and McIntyre 1995, Schlosser 1995, Ward 1998), but these efforts focused on large spatial scales (e.g., riverine landscapes) and moderately long time scales (e.g., generations). Effects of fragmentation on fish behavior (e.g., foraging, habitat selection, and response to disturbance) remain poorly understood over short time periods and at small spatial scales.

Riffles are typically the corridors connecting neighboring pools in stream systems with well-developed habitats. Characteristics of riffles (e.g., shallow depths, fast currents) may limit the ability of pool dwelling fishes to access neighboring pools. Access to neighboring pools may be particularly limited if pools are infrequent or spacing of pools is increased in association with habitat degradation. Although riffles may not preclude all fish movement, long riffles may significantly reduce excursions between habitat patches, slow immigration to new habitats, and limit the ability of fish to track variability in food resources and predator densities.

Effects of Isolation on Response to Disturbances

Episodic disturbances, such as floods and droughts, are major factors shaping the organization of stream communities (e.g., Pearsons and others 1992, Strange and others 1992). The rate at which assemblages recover from disturbance will depend strongly on how rapidly different fish species recolonize disturbed stream segments. A large body of evidence indicates that recolonization of disturbed stream segments by fishes can be fairly rapid (< 1 yr, e.g., Niemi and others 1990). Recovery of fish assemblages in short reaches and habitat units can occur on the scale of days and weeks (Peterson and Bayley 1993, Sheldon and Meffe 1995). Nevertheless, research thus far has revealed little about physical factors that contribute to recovery, especially at small spatial scales (Detenbeck and others 1992). For example, is recolonization influenced by the size of the habitats affected or by the degree to which affected habitats are isolated from source habitats as predicted by Island Biogeography Theory (MacArthur and Wilson 1967).

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Effects of Isolation on Short Term Movement

Biologists have long been interested in the movement of fish in streams though few have studied how habitats, when organized into discrete patches, may affect movement behavior. Gerking (1953) was probably the first to discuss the possible effects of patchiness when he proposed that long riffles separating adjacent pool habitats could act as behavioral barriers to movement and that fishes within individual pools could be viewed as discrete populations. Matthews and others (1994) also speculated that riffles act as size selective barriers to movement and affect pool assemblage dynamics. Recent experiments added weight to earlier observations by showing that long riffles slow fish recolonization to defaunated pools (Lonzarich and others 1998) and reduce fish movement between pools in experimental streams (Schaeffer 1999).

One message that emerges from field and experimental studies is that the spatial mosaic of stream habitats may significantly influence the short term movements of fish. At least with respect to short time scales (i.e., days, weeks), the spatial arrangement of pools and riffles in streams may have significant ecological implications. For example, pool dwelling fish may need to traverse long, shallow riffles to access suitable habitats. Depending on the distance between adjacent patches (i.e., length of intervening riffles), these movements may entail high energetic costs or high predation risks (e.g., Power 1987).

We strongly believe that efforts to better understand the ecology of stream fishes and more specifically, the consequences of land use activities, will benefit from research aimed at addressing the effects of habitat patchiness on fish movement. In this paper, we summarize the major findings and discuss the general implications of two different, but closely related, studies that examined the effects of habitat isolation on recolonization and daily movement patterns of pool dwelling fishes in two Arkansas streams. Complete descriptions of the results of both experiments can be found elsewhere (Lonzarich and others 1998, Lonzarich and others, in press).

METHODS

Study Area

We carried out the experiments in the summers of 1995 and 1997 in two tributaries of the Little Missouri River in the Ouachita National Forest, west central Arkansas (34°22'30"lat and 93°52'30"long). Long and Blaylock creeks are short (< 10 km), moderate gradient systems that flow through predominantly forested and mountainous terrain. General characteristics of the streams included bedrock and gravel substrates and dense riparian vegetation. The proportion and average size of pool and riffle habitats in the two streams were similar (Clingenpeel 1994) as were the composition, rank abundance, and densities of fish species (Lonzarich and others 1998).

Experimental Design

General features—We selected pools from the two streams and assigned them equally into two treatment categories based primarily on differences in the lengths of adjacent riffles. Short riffle pools were separated from adjacent upstream and downstream pools by riffles < 10-m long; long riffle pools were separated from adjacent pools by riffles which averaged ≥ 50 m in length. With the exception of these differences, the physical dimensions of pools in the two treatment groups were similar (Lonzarich and others 1998, Lonzarich and others, in press). The recolonization experiment included 12 treatment pools (6 pools per treatment), and the movement experiment included 16 pools (8 pools per treatment).

In both experiments, we collected fishes by isolating survey pools with 6-mm mesh block nets and sampling with a Smith Root battery powered, backpack electrofisher. For the recolonization experiment, we sacrificed collected fish or relocated them downstream of barriers to upstream movement. For the movement study, we marked collected fish with treatment specific caudal fin clips (i.e., long riffle pools, upper caudal fin; short riffle pools, lower caudal fin).

Species composition, rank dominance, and fish densities in Long and Blaylock creeks were very similar (Lonzarich and others 1998). In both experiments, we focused on a subset of fish species. Because of concerns over sampling efficiency, we did not include small, juvenile fish (< 25 mm) or bottom dwelling species that often hide within the substrate and can be difficult to locate without considerably increasing survey times. We excluded four bottom dwelling species from the survey: northern hog sucker, *Hypentelium nigricans* (Lesueur); orangebelly darter, *Etheostoma radiosum* (Hubbs and Black); greenside darter, *E. blennioides* Rafinesque; and yellow bullhead, *Ameiurus natalis* (Lesueur). Numerically, these species and small juveniles were a minor component of the pool assemblages, accounting for less than 10 percent of all fish collected by electrofishing (Lonzarich and others 1998). We included seven target species in the surveys: central stoneroller, *Campostoma anomalum* (Rafinesque); striped shiner, *Luxilus chrysocephalus* (Rafinesque); redfin shiner, *Lythrurus umbratilis* (Girard); bigeye shiner, *Notropis boops* Gilbert; northern studfish, *Fundulus catenatus* (Storer); creek chub, *Semotilus atromaculatus* (Mitchill); longear sunfish, *Lepomis megalotis* (Rafinesque); and smallmouth bass, *Micropterus dolomieui* Lacepede.

We quantified both recolonization and short term movement using underwater observation rather than electrofishing largely because our sampling design required repeated fish counts over short periods. We previously showed that population estimates generated from snorkeling surveys are nearly identical to those generated from more invasive and labor intensive electrofishing surveys (Lonzarich and others 1998). High water clarity in each stream provided excellent conditions for identifying marked fish from distances up to 3 m.

Recolonization protocol—We censused fish in experimental pools by snorkeling immediately prior to electrofishing (Day 0, predisturbance census), and then 1, 3, 10, 20, 30, and 40 days following the removal of target species. To minimize observer error, the same person surveyed all pools, and two consecutive censuses were performed for each survey. The mean of these two counts was used in statistical analyses.

For this summary, we limited characterization of recovery patterns for species and entire pool assemblage to changes in the relative abundance of fish within each pool. We divided counts (individuals per pool) obtained for any post disturbance census by predisturbance counts from Day 0 so that a relative abundance of 100 percent indicated complete recovery. Using relative abundance data for each survey date, we generated regression models to characterize recovery patterns and rates for each treatment group. The model that best explained the relationship between time and recovery had the following linear form:

$$ln(Recovery) = ln(a) + b*(ln(Day + 1)) \qquad (1)$$

where recovery is either assemblage or species recovery (relative numbers), and b, the regression coefficient, represented the recovery rate. We used a two sample t test to compare differences in mean assemblage and species recovery rates (b) in the two treatment groups. We determined the effects of other independent variables (i.e., riffle depth, distance to large pool, pool area and assemblage size) on pool specific recovery rates by least squares regression. If percentage data deviated from normality, we performed square root arcsine transformations prior to analysis.

Movement protocol—We marked nearly 1,900 individuals from the 16 experimental pools. After marking, we held all fish overnight and released them into their experimental pools 30 min before we initiated snorkeling surveys. In upstream and downstream pools immediately adjacent to the experimental pool, we completed simultaneous surveys hourly between 1100 to 1700 h (6 censuses) on Day 0 and once per day on Day 1 and Day 3 between 1100 to 1400 h for a total of 8 censuses of marked fish. On Day 1 and Day 3, our snorkeling censuses included the experimental pool and the second upstream pool and second downstream pool in the study reach. On Day 0, we limited censuses of the experimental pool to the final survey (1700 h) to minimize the risk that snorkeling might artificially inflate emigration.

An important assumption of our study was that marked fish observed in upstream and downstream pools accurately reflected emigration from experimental pools and were not biased downwards by fish emigrating from an experimental pool to become established in adjacent riffles. We accepted this assumption on the basis of previous research where we found strong differences in pool and riffle assemblages of these streams (Lonzarich and others 1998).

We estimated the rate, direction, and percentage of emigrants from each pool. We determined movement rates (fish per h) by least squares linear regression models using data from Day 0 hourly surveys. Because the general shape of the relationship between time and movement was logarithmic, the model showing the best fit to the data had the following general form:

$$Movement = a + b*ln(h + 1) \qquad (2)$$

where movement represented the percentage of marked fish observed outside of the experimental pool and b, the regression coefficient, represented the movement rate. We generated species specific and assemblage level linear regression models for each experimental pool. To test for treatment differences in the rate of movement, we compared mean movement rates (b) for each treatment using a two sample t test. We determined the directionality of movement by dividing the number of marked fish that moved downstream by the combined number of emigrants. Hence, values above 50 percent indicated an upstream bias in movement and values below 50 percent, a downstream bias. We computed the percentage of emigrants from each pool by dividing the maximum number of known emigrants by the total number of fish marked in the experimental pool. We typically observed the maximum number of emigrants on Day 3 in long riffle pools and on Day 1 in short riffle pools. We tested for species and assemblage level differences in movement patterns (i.e., rate, proportion, and direction) between the two treatment groups using two sample t tests (rate and proportion) and 2 x 2 Chi square tests of independence (for direction). If percentage data deviated from normality, we performed square root arcsine transformations prior to analysis.

RESULTS

Effects of Isolation on Recolonization

Numerical recovery was influenced strongly by factors that isolated experimental pools from potential colonists. In short riffle pools, assemblages reached full numerical recovery by day 30, whereas assemblages in long riffle pools reached only 75 percent of their predisturbance densities by day 40 (fig. 1). Estimates of full recovery based on data extrapolation ranged from 100 to 130 d for long riffle pools. Over the entire recovery period, the mean (± 1 S.E.) recovery rate in long riffle pools was 33 percent lower (20 ± 0.6 percent per ln(d)) than the rate for short riffle pools (30 ± 1.5 percent per ln(d), t test, $P < 0.05$). In addition, recovery was correlated positively with riffle depth ($r^2 = 0.37$, $P < 0.05$, fig. 2a) and negatively with distance to the nearest large source pool ($r^2 = 0.44$, $P < 0.05$, fig. 2b). These patterns held even when pools within each riffle length category were examined separately. Neither predisturbance assemblage size nor pool area had any detectable influence over numerical recovery rates of assemblages ($P > 0.50$, fig. 2c).

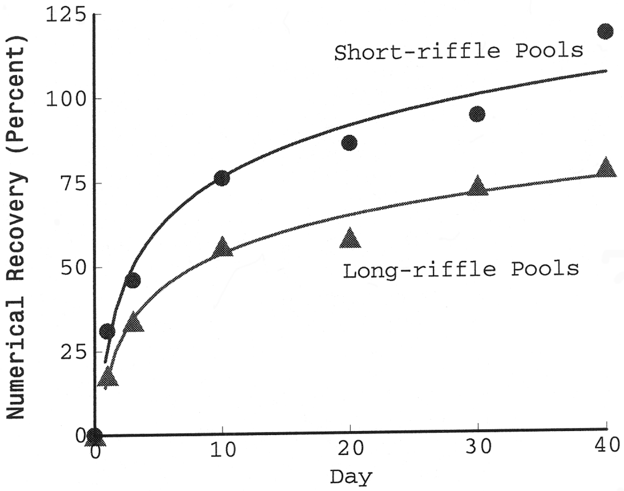

Figure 1—Comparisons of percent numerical recovery for each riffle length category. Error bars represent ± 1 S. E. Data points and error bars are offset to increase clarity.

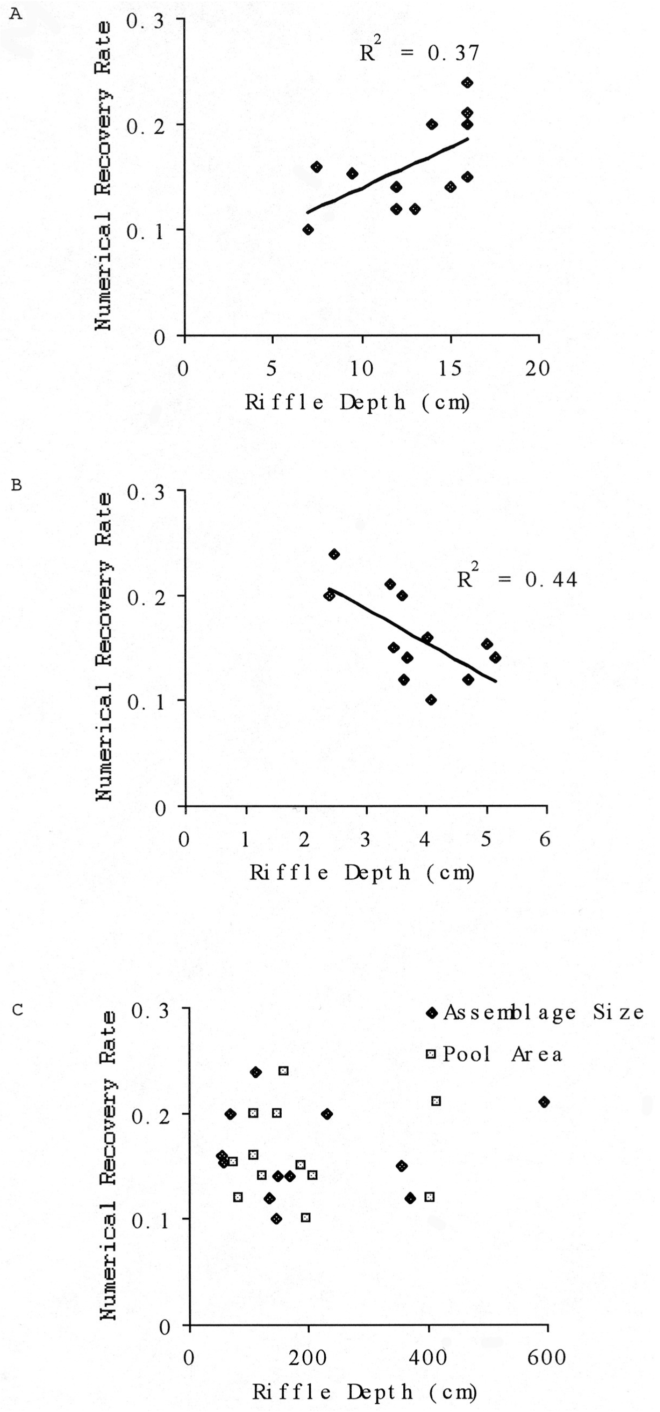

Figure 2—Relationships between estimated recovery rates (A) riffle depth, (B) Ln distance to large pool, and (C) predisturbance assemblage size and experimental pool area.

Effects of Isolation on Short Term Movement

Marked fish emigrated from short riffle pools much more frequently (up to 10 fold) than from long riffle pools. The emigration rate (± 1 S.E.) from short riffle pools on Day 0 was seven times higher ($b = 6.9 \pm 0.7$ percent per ln(h) than the estimated rate for long riffle pools ($b = 1.0 \pm 0.3$ percent per ln(h), t test, $P < 0.01$). In short riffle pools, we commonly observed emigrants during the first count, while in long riffle sites, we did not see emigrants until at least the third hour and generally observed very few on Day 0 (< 2 percent). By Day 3, we observed an average of 30 percent (1 SE = ± 3.8 percent) of marked fish outside of experimental short riffle pools. In contrast, the average for long riffle pools was only 9 percent (1 SE = ± 2.3 percent).

When we account for all marked fish observed within each site, differences in emigration between treatments become even more trenchant. Expressed as a percentage of the total number marked, the proportion of marked fish observed within long riffle reaches did not change over time (fig. 3). In contrast, the percentage of marked fish declined sharply in short riffle reaches from 67 percent for Day 0 to 49 percent for Day 3. Assuming that all of these individuals (18 percent) moved beyond the study area boundaries by Day 3, the actual percentage of fish emigrating from short riffle pools was much higher than the observed average emigration of 30 percent.

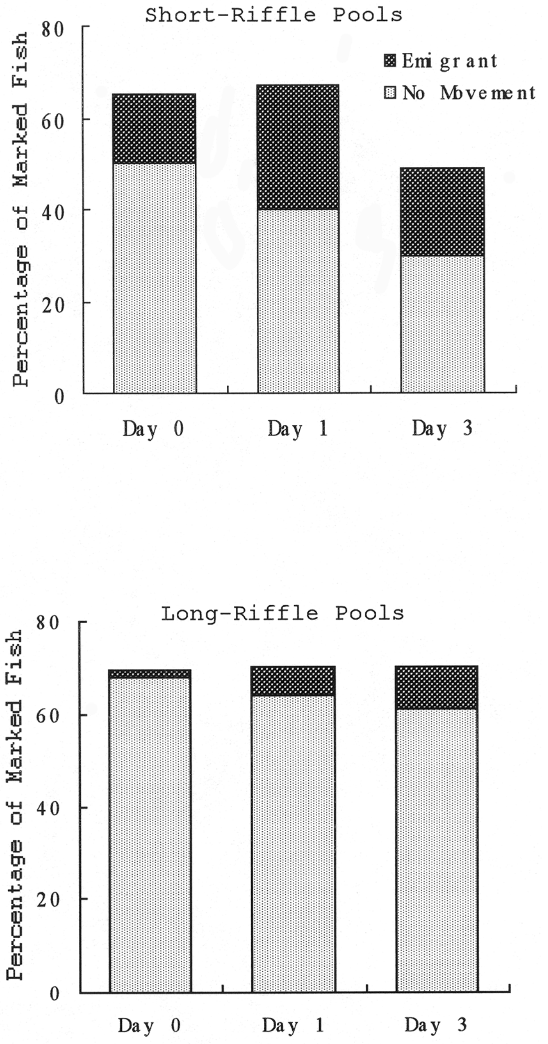

Figure 3—Marked fish (percent) emigrating from and remaining within experimental pools on each day of the three-day survey periods.

Marked fish moving from short riffle pools selected upstream and downstream directions with equal frequency (52 vs. 48 percent, respectively). In contrast, emigrants from long riffle pools moved downstream nearly twice as often as they moved upstream (63 percent vs. 37 percent, respectively, Chi square goodness of fit test, $P < 0.01$).

DISCUSSION

We summarized here the first studies to quantify how the habitat mosaic in streams can shape the response of fish to acute disturbances and limit their short term movements among habitats. Attributes of the physical environment responsible for this variability in movement behavior included riffle length, depth, and distance to large source pools.

Though poorly studied in streams, the effects of habitat isolation on the movement of organisms (e.g., Forman and Godron 1986, Forman 1995) and on the recolonization of habitat islands (e.g., Simberloff and Wilson 1969) have long been recognized by ecologists. In fact, Brown and Brussack (1990) described riffle habitats for benthic invertebrates as isolated islands that were separated by long pools of low habitat quality. Like organisms in other spatially heterogeneous environments, fish in streams are distributed within a mosaic of habitats of varying quality (e.g., resource availability, predator densities, and physical conditions). Thus, the rate at which individuals move between patches and recolonize disturbed reaches should depend on whether intervening habitats (i.e., riffles) possess properties that could impair movement. For pool dwelling species, we suspect that limited movement through long riffles may be due, at least in part, to the fact that these habitats are shallow and fast flowing. Fast flows may be difficult to negotiate, and shallow water may expose individuals to terrestrial predators (e.g., Power 1987). Although direct field evidence supporting either of these potential factors is lacking, Warren and Pardew (1998) measured a strong effect of water velocity on fish movement through road crossings, and we found that fish in long riffle pools moved significantly less frequently against the current (upstream) than with the current (downstream). We also measured a negative relationship between water depth and fish recolonization of defaunated stream pools (Lonzarich and others 1998).

We suspect that these factors contribute to variability in fish movement through long riffles but urge care in extrapolating beyond the temporal scope (i.e., days and months) and low flow conditions that characterized our experiments. We actually question whether potential survival or energy costs associated with traversing long riffles affect patterns of fish redistribution during floods or seasonal migrations to spawning or wintering habitats. Nevertheless, the timing of our study, the summer, coincided with a critical period in the life cycles of stream fish species. Not only is this the season when fish experience their most favorable growing conditions, it also is when competition and threats of predation, especially for juveniles, can be very intense. Further, in Arkansas, as in many other geographic regions, stream organisms are often exposed to unpredictable and potentially severe flow disturbances during the summer.

Given this context, we ask how variability in fish movement between pools affects the dynamics of populations and assemblages both at the scale of habitats and stream reaches? As reflected in our study design, we believe that there are at least two ways to approach this question. When measured over relatively long time periods (e.g., days, months), it appears that pool isolation can severely limit the rate at which fish assemblages recover from acute disturbances as predicted by Island Biogeography Theory (MacArthur and Wilson 1967). Not only is recovery affected by the distance to potential colonists, it also is affected by the proximity of large pools, which in some respects may be analogous to the mainland habitats of MacArthur and Wilson (1967). Sedell and others (1990) argued that large pools are very important to the ecology of streams because they provide stream organisms with refugia during stressful conditions (e.g., droughts, floods). Our results imply that large pools, which were two times larger than average sized pools in our study streams, also may serve as important sources of immigrants to neighboring up and downstream habitats. In view of the importance of episodic disturbances in streams, and because land use activities can alter both the stream flow regime (e.g., Rosén and others 1996) and habitat spacing, these results may help explain variability in the organization of fish assemblages in stream pools.

When viewed over much shorter time scales (e.g., hours, days), pool isolation may affect the ability of fish to track spatially variable resources (e.g., food, mates) or predator densities. Matthews (1998) described such a possibility. He speculated that differences in the ability of small bodied prey and large bodied predators to traverse shallow riffles might affect predator prey relationships in stream reaches. The assumption of open fish communities in pools appears to conflict with the restricted movement paradigm (Gerking 1953) and with the widely held view that fish within pools can be viewed as discrete populations (e.g., Gerking 1953, Matthews and others 1994). However, it is consistent with the findings of Fraser and Sise (1980) who proposed that stream fish should be sensitive to local resource availability and move between pools in search of food resources. We add that the possible benefits of moving from a pool must be weighed against the potential costs of crossing riffles. Therefore, in long riffle pools where movement can be limited, there may be a tendency towards closed community structure with local (i.e., pool scale) conditions, such as prey abundance, predator densities, and the habitat template having strong affects on assemblage organization. In short riffle pools, where the daily turnover of individuals can be very high (> 20 percent per day), the characteristics of pool assemblages and populations may show little if any relationship to local conditions. This view is similar to that of Cooper and others (1990) who found that the ability of fish predators to control the local abundance of stream insects diminished as insect exchange rates (emigration/immigration) into pools increased.

SUMMARY

The extent and nature of animal movements are key factors affecting the vulnerability of species to landscape changes (Law and Dickman 1998). Historically, stream ecologists have focused on local habitat conditions (e.g., pools, riffles) when trying to describe and predict the factors that explain the abundance and dynamics of fish populations. The importance of habitat isolation to the ecology of stream fishes may vary widely across streams and geographic regions due to variability in the physical factors that determine channel features. Riffle lengths in our streams were probably representative of conditions in many small upland stream systems in the eastern United States but are much shorter than those found in small montane streams of the Pacific Northwest (e.g., > 80 m, Beechie and Sibley 1997). In contrast, low gradient, sand bottom streams of the Gulf and Atlantic Coastal Plains may lack physical conditions promoting riffle development. Logging, agriculture, and other land use activities along streams can lead to changes in the channel course or reduce the availability of instream elements (e.g., large woody debris) that create habitat. Such changes can reduce the abundance, and therefore the spacing, of pools in streams. For this reason, we strongly recommend that efforts to understand the ecology of stream fishes and the consequences of land use activities focus more attention on the spatial distribution and potential isolation of these habitats in streams.

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USING SNORKELING TO QUANTIFY FISH ASSEMBLAGE STRUCTURE IN ARKANSAS STREAMS

David G. Lonzarich, Mary E. Lonzarich, and Melvin L. Warren, Jr.[1]

Abstract—As a technique for surveying fish populations in low diversity, clear streams of the Pacific Northwest, snorkeling is a commonly used alternative to electrofishing. While the method is becoming more widely used in the high diversity streams of eastern North America, its efficacy as a tool for surveying fish assemblages in these streams is not known. In this study, we compared results from snorkeling and electrofishing surveys obtained from two clear, warmwater streams of the Ouachita Mountains, Arkansas. Surveying only water column species, we found that fish assemblage structure and species density estimates obtained by the two methods were statistically indistinguishable. These results have important implications for ecologists studying warmwater streams as snorkeling holds many advantages over invasive sampling methods, such as seining and electrofishing.

INTRODUCTION

When planning fish surveys, stream ecologists can choose from a variety sampling techniques, although seining, electroshocking, and underwater observation are the most widely used. Seining and electroshocking are both invasive sampling methods that traditionally have been favored in most stream surveys, especially in studies that do not require repeated measurements (e.g., collections of baseline population data, microhabitat use or feeding). Unfortunately, as with all sampling methods, seining and electrofishing possess potentially severe limitations. For example, habitat conditions can dramatically affect seining success. In streams with uneven bottoms and physical obstacles, such as large rocks, vegetation or woody debris, seining efficiency can be very low. Electrofishing is more effective under these conditions, but it too can suffer limited efficiency (e.g., Riley and Fausch 1992). Importantly, electrofishing can injure or kill collected individuals, an especially serious problem in waters with endangered or recreationally important fishes (Nielsen 1998). Finally, all invasive methods require the handling of individuals to retrieve necessary data. At worst, handling can cause injuries or death; at best, it may affect the behaviors of sampled individuals.

In many situations, snorkeling may be a solution to the limitations of invasive sampling. Snorkeling, which has advantages of versatility and cost effectiveness, can provide information on the composition, distribution, abundance, and behavior of fishes in streams (Dolloff and others 1996). Snorkeling is widely used in small, clear streams of the Pacific Northwest, especially in studies of trout and salmon (e.g., Hankin and Reeves 1988, Hillman and others 1992). Snorkeling provides several major benefits. A variety of habitat conditions can be surveyed in relatively short periods of time, and labor and equipment costs are low compared to other techniques. Importantly, fishes are not collected or handled in snorkeling surveys. Hence, studies requiring repeated observations (e.g., movement studies) are not compromised by handling stress, which can adversely affect behaviors of interest to the researcher and the experimental results (Lonzarich and others 1998, Lonzarich and others, in press). Moreover, survey efficiency is not impeded by physical factors such as wood, undercut banks, or uneven substrates. A major drawback to snorkeling is the requirement for clear waters with very high visibility > 2 m, (Whitworth and Schmidt 1980). Depending on the study objectives, snorkeling surveys also may require specialized training of observers to assure accurate, underwater identification of fishes.

Several studies have assessed the efficacy of snorkeling in low diversity, clear streams of the Pacific Northwest. The method can yield very precise results (e.g., Hillman and others 1992, Dolloff and others 1996), but efforts to assess snorkeling accuracy are difficult because true population densities are usually unknown (Dolloff and others 1996). In the more species rich streams of eastern North America, the precision and accuracy of this technique as a tool to survey fish assemblages are poorly documented. In two other studies, we monitored changes in the population and assemblage structure of stream pools. In conjunction with that research, we report here a study conducted to assess the effectiveness of snorkeling as a census method in streams of the Ouachita Mountains, Arkansas. Surveying water column fish species in two moderately diverse streams, our objective was to compare estimates of population density and assemblage structure from snorkeling with results obtained by backpack electrofishing.

METHODS

We conducted electrofishing and snorkeling surveys between June and July 1995 in two tributaries of the Little Missouri River in the Ouachita National Forest, Arkansas (34°22′30″ lat and 93°52′30″long). Long and Blaylock creeks are relatively short (< 10 km), low gradient systems that flow through forested and mountainous terrain. General characteristics of the streams included bedrock, cobble, and gravel substrates and dense riparian vegetation. Using habitat inventory data (Clingenpeel 1994) and data from our own surveys, we selected 12 pools in the two streams. For each

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pool, we measured total length, width, area, maximum depth, and substrate composition (i.e., boulder, cobble, and gravel). We also measured biological characteristics including species richness, total assemblage densities, and individual species densities.

Fish Surveys

Fish species composition, rank dominance, and densities in Long and Blaylock creeks are very similar (Lonzarich and others 1998). In this study, we focused on a subset of fish species. Because of concerns over sampling efficiency, we did not include small, juvenile fish (< 25 mm) or bottom dwelling species that often hide within the substrate and can be difficult to locate without considerably increasing survey times. We excluded four bottom dwelling species from the survey: northern hog sucker, *Hypentelium nigricans* (Lesueur); orangebelly darter, *Etheostoma radiosum* (Hubbs and Black); greenside darter, *E. blennioides* Rafinesque; and yellow bullhead, *Ameiurus natalis* (Lesueur). Numerically, these species and small juveniles were a minor component of the pool assemblages, accounting for less than 10 percent of all fish collected by electrofishing (Lonzarich and others 1998). We included seven target species in the surveys: central stoneroller, *Campostoma anomalum* (Rafinesque); striped shiner, *Luxilus chrysocephalus* (Rafinesque); redfin shiner, *Lythrurus umbratilis* (Girard); bigeye shiner, *Notropis boops* Gilbert; northern studfish, *Fundulus catenatus* (Storer); creek chub, *Semotilus atromaculatus* (Mitchill); longear sunfish, *Lepomis megalotis* (Rafinesque); and smallmouth bass, *Micropterus dolomieui* Lacepede.

For each survey, we first snorkeled and then immediately electrofished the study pool. We began snorkeling surveys at the downstream end of the pool, continued in a zig-zag fashion to the upstream end of the pool, and then snorkeled downstream. We considered this a single pass, which generally took less than 30 min to complete. We counted fish individually except when fish were aggregated; for aggregations, we estimated numbers by counting individuals in groups of 5 to 10 individuals. To minimize observer error, the same observer conducted surveys of all pools censused. Each survey consisted of two passes with the average of the two counts used in all statistical analyses.

Immediately after the snorkeling survey, we electrofished experimental pools, isolating them with 6-mm mesh block seines and sampling with a Smith Root battery powered, backpack electrofisher. We sampled pools four to seven times until no fish were collected on two consecutive passes. Because of this intensive sampling effort and because we identified fish in the field, surveys of individual pools took 3 to 4 h to complete. We evaluated the effectiveness of the electrofishing technique in removing all target species by conducting snorkeling surveys in the isolated pools immediately upon the completion of sampling. In post-electrofishing snorkeling surveys of 6 of the 12 pools, we observed only 3 fishes from the target group of species.

Data Analysis

We estimated the efficiency of snorkeling relative to electrofishing. To do so, we compared assemblage structure (i.e., species richness, percent similarity, and total numbers) and species abundance (fish per m^2) results from snorkeling surveys with results from electrofishing surveys. We compared assemblage similarity using the Percent Similarity Index (Wolda 1981). We used correlation analysis to evaluate the strength of relationships between the two methods. When data met the assumption of normality, we used the Pearson Product Moment analysis. Otherwise, we used Spearman Rank analysis. We compared species and assemblage density estimates for the two methods by a two-sample t-test ($P < 0.05$).

RESULTS

Snorkeling surveys yielded results that were very comparable to those obtained in electrofishing surveys. With respect to the total number of fish per pool, estimates for the two methods were highly correlated (Pearson product moment correlation coefficient, $r^2 = 0.98$, n = 12, fig. 1). On average, total counts from snorkeling observations were within 10 percent of the total counts from electrofishing surveys (1.03 fish per m^2 versus 1.13 fish per m^2, respectively). Neither size of pool nor size of pool assemblages (range, 63 to 656 individuals per pool) affected this level of precision.

Estimates of assemblage structure (species composition, rank abundance, and relative abundance) for the two methods also were very similar. Across the 12 pools, we missed species in snorkeling surveys that were captured by electrofishing on only two occasions (out of 82 possibilities). Similarly, on a single occasion, we missed only one species in electrofishing surveys that we observed snorkeling. Correspondence in the rank abundance of species for the two methods was very high (Spearman Rank correlation coefficient, average = 0.97; range, 0.87 to 1.00). The relative abundances of species for each method also were very

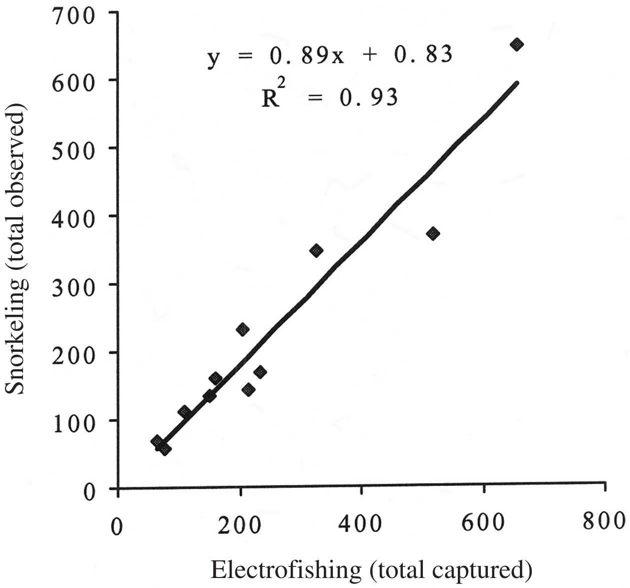

Figure 1—The relationship between electrofishing and snorkeling estimates (total counts per pool) for 12 experimental pools in Long and Blaylock Creeks, Ouachita Mountains, Arkansas.

Table 1—Comparison of mean fish densities[a] estimated from electrofishing and snorkeling surveys of 12 treatment pools in Long and Blaylock Creeks, Ouachita Mountains, Arkansas

Species	Snorkeling	Electrofishing	Difference
			Percent
Central stoneroller	50.5 ± 13.4	51.6 ± 10.8	2
Striped shiner	30.0 ± 6.1	30.0 ± 6.5	0
Longear sunfish	7.4 ± 1.6	10.4 ± 2.3	29
Creek chub	7.5 ± 3.2	9.3 ± 4.1	19
Northern studfish	2.5 ± 0.7	5.4 ± 1.7	54
Redfin shiner	2.3 ± 0.8	2.0 ± 0.6	15
Bigeye shiner	1.1 ± 0.6	1.9 ± 1.1	42
Smallmouth bass	1.6 ± 0.4	1.9 ± 0.5	16
All species	103.2 ± 14.5	113.2 ± 12.6	8

[a] Individuals/100 m^2 ± 1 standard error of the mean.

comparable, with an average assemblage similarity of 91 percent across the 12 pools (range, 84 to 94 percent).

Snorkeling proved to be nearly as effective as electrofishing in estimating species densities. On average, snorkeling estimates for the eight target species were slightly lower than electrofishing densities (table 1). Snorkeling efficiency was not related to abundance. Estimates for the two most common species, central stoneroller, and striped shiner, were nearly identical to electrofishing estimates (table 1). The weakest relationships were for northern studfish (54 percent deviation), longear sunfish (29 percent deviation) and bigeye shiner (42 percent deviation). However, neither these nor any other snorkeling and electrofishing estimates of species densities were significantly different (t-test, $P >$ 0.25, table 1). We attribute the northern studfish results to our failure to recognize their preference for shallow, marginal habitats in early surveys. Our efficiency likely improved in later surveys as we became more aware of their patterns of habitat use. We believe the longear sunfish results likely reflect their use of cover in the presence of a snorkeler, whereas the bigeye shiner finding is related to their rarity in the study pools.

DISCUSSION

Because we removed all target fish species from experimental pools during electrofishing surveys, we were able to simultaneously compare the effectiveness of snorkeling against electrofishing and to determine its accuracy. At least for the water column species targeted, our results show that snorkeling is a very accurate method for characterizing fish assemblage and species abundance patterns in small, clear warmwater streams. Our findings have several important implications. First, snorkeling required much less survey time (0.5 h versus 3 to 4 h) and labor than electrofishing. Hence, more area can be sampled by snorkeling than by electrofishing or seining. Second, snorkeling, unlike electrofishing, does not adversely affect surveyed fishes and can be used to describe short-term changes (e.g., days, weeks) in populations and assemblages. As we demonstrated elsewhere (Lonzarich and others 1998) (Lonzarich, D.G.; Lonzarich, M.R.; Warren, M.L., Jr. Effects of riffle length on the short-term movement of fishes among stream pools. Manuscript in preparation.) repeated snorkeling surveys can be used to quantify the daily movements of fish and their patterns of recolonization following disturbance. Recently, we repeatedly snorkeled pools in one Arkansas stream over the course of a summer to determine pool specific patterns of extinction and colonization of water column species (Lonzarich, D.G.; Lonzarich, M.R.; Warren, M.L., Jr. Effects of riffle length on the short-term movement of fishes among stream pools. Unpublished data. On file with: D. Lonzarich, University of Wisconsin, Eau Claire, Department of Biology, Eau Claire, WI 54702). Previously in this same region, Matthews and others (1994) used snorkeling to repeatedly census fishes from 14 pools over a 19-month period.

While advocating snorkeling surveys for selective fisheries applications, we are mindful of its many limitations. For example, although benthic species (e.g., darters, catfish, suckers) can be censused in snorkeling surveys, they likely cannot be surveyed by snorkeling as efficiently as water column species. Further, inclusion of benthic fishes would probably add significantly to the time needed to complete an assemblage survey. Nonetheless, in studies of clear streams, where it is not necessary (and possibly detrimental) to handle fish, the technique has important advantages over invasive sampling methods.

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COMPUTERIZATION OF THE ARKANSAS FISHES DATABASE

Henry W. Robison, L. Gayle Henderson, Melvin L. Warren, Jr., and Janet S. Rader[1]

Abstract—Until recently, distributional data for the fishes of Arkansas existed in the form of museum records, field notebooks of various ichthyologists, and published fish survey data; none of which was in a digital format. In 1995, a relational database system was used to design a PC platform data entry module for the capture of information on the fishes of Arkansas. The graphical user interface (GUI) consists of four screens for data entry of information on locality, habitat, collection details, and fish species. Values for most of the fields in the data entry screens were standardized to reduce input error and ensure consistency. Look-up tables allow the user to quickly select standardized values (e.g., fish species and family, river and stream systems, county, habitat) for many fields. Comment fields were incorporated to allow the user to record important information on individual specimens that was not amenable to standardization. The database is fully compatible with geographic information systems and each collection in the database is geo-referenced. The computerization of this large database of fishes was initiated as part of a cooperative effort with the USDA Forest Service to digitally capture historic information on fishes of the Ouachita National Forest. The agreement was later expanded to include fish records from the Ozark and St. Francis National Forests. Recently, records of fishes from the entire state were added to the database with funding from the Arkansas Game and Fish Commission. Currently, over 3,500 fish collections have been entered into the Arkansas Fishes Database.

BACKGROUND

In 1994 the first author, Henry W. Robison, and his staff completed the first tests of a data entry module to capture recent and historical fish distributional information on Arkansas fishes that existed only in manual files. The second author, L. Gayle Henderson, developed the module with assistance from the third author, Melvin L. Warren, Jr. The pilot study was completed in September 1995, and the module proved to be time-efficient and flexible. Data entry of Robison's Arkansas fish collections was begun in the fall of 1995 under a Challenge Cost-Share (CCS) agreement with the Ouachita National Forest (ONF), Southern Research Station, and The Nature Conservancy. This CCS agreement was extended later in partnership with The Nature Conservancy, the ONF Ecosystem Management Program, the ONF Fisheries Program, and Robison.

Data entry proceeded with records of fishes from within the proclamation boundaries of the ONF. Initially, the records entered were those of Robison; however, permission was secured subsequently from Neil H. Douglas, Northeast Louisiana University (NLU), to add the data from fish collections housed in their museum. This data was primarily from the fieldwork of Douglas and NLU graduate students. Later, data on Arkansas fishes from the large holdings of the Tulane University fish museum were added to the fish database.

The USDA Forest Service effort focused primarily on fishes living within or immediately downstream of the forest proclamation boundaries. Initial data entry concentrated on that subset of fish collections but also included collections from river systems in Arkansas that drain the ONF. In a subsequent CCS in 1997, fish collection data from the proclamation boundaries of the Ozark National Forest and the St. Francis National Forest were added to the expanding fish database. More recently, information from field collections made by Tom M. Buchanan, John L. Harris, Betty Crump, George L. Harp, and others in the national forest areas and elsewhere in Arkansas were added. The addition of other national forests and field collections of these individuals substantially increased coverage of federal and surrounding lands in Arkansas.

In 1998-99, supplemental funding was granted from the Arkansas Game and Fish Commission AGFC to include fish collections from all drainages in Arkansas. The result is a database covering the entire state of Arkansas which can be used by the USDA Forest Service, The Nature Conservancy, the AGFC, the Department of Arkansas Natural Heritage, and others in future planning, monitoring, and management efforts.

PERTINENT LITERATURE

The site-specific information now contained in the fishes of Arkansas database formed the basis of several scientific contributions that increased our knowledge of fish distribution, fish conservation status, and fish assemblage association with watershed characteristics. Robison and Buchanan (1988) published "Fishes of Arkansas" with dot distribution maps depicting over 3,000 fish collections within the state. These localities were located by hand on paper maps that are currently in the possession of Robison. None of the more than 3,000 collections was in a digital format, decreasing their utility for rapid manipulation, analysis, planning, and monitoring. This manually compiled information was later used by Matthews and Robison (1988), Matthews and others (1992), and Matthews and Robison (1998) for studies analyzing the distributions of Arkansas fishes and the geological, climatological, and water quality correlates that described faunal patterns across the state. Recently, the

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maps of Robison and Buchanan (1988), in part, formed the basis for an assessment of fish distribution, diversity, and conservation status for hydrologic units in the Ozark-Ouachita Highlands Assessment led by the USDA Forest Service (Standage 1999; Warren and Clingenpeel 1999; Warren and Hlass 1999; Warren and Tinkle 1999).

OBJECTIVES

The computerization of the distributional data for the fishes of Arkansas was premised on information needs in four specific areas. First, there was a need to establish a Geographic Information System (GIS)-compatible fish research database for the state of Arkansas to document historical and present fish distributions. Second, a digital database would allow identification of unique ecological or taxonomic fish community assemblages and centers of fish diversity within and across drainages of Arkansas. Third, historical changes in stability and persistence of community assemblage patterns and historical trends in species distributions needed to be easily associated with land use. Finally, there was a need for geo-spatial tools to assess conservation status of individual Arkansas fish species. The primary objective of this paper is to describe the development and design of the database used to capture collection records for fishes across Arkansas and that can be used to meet these information needs.

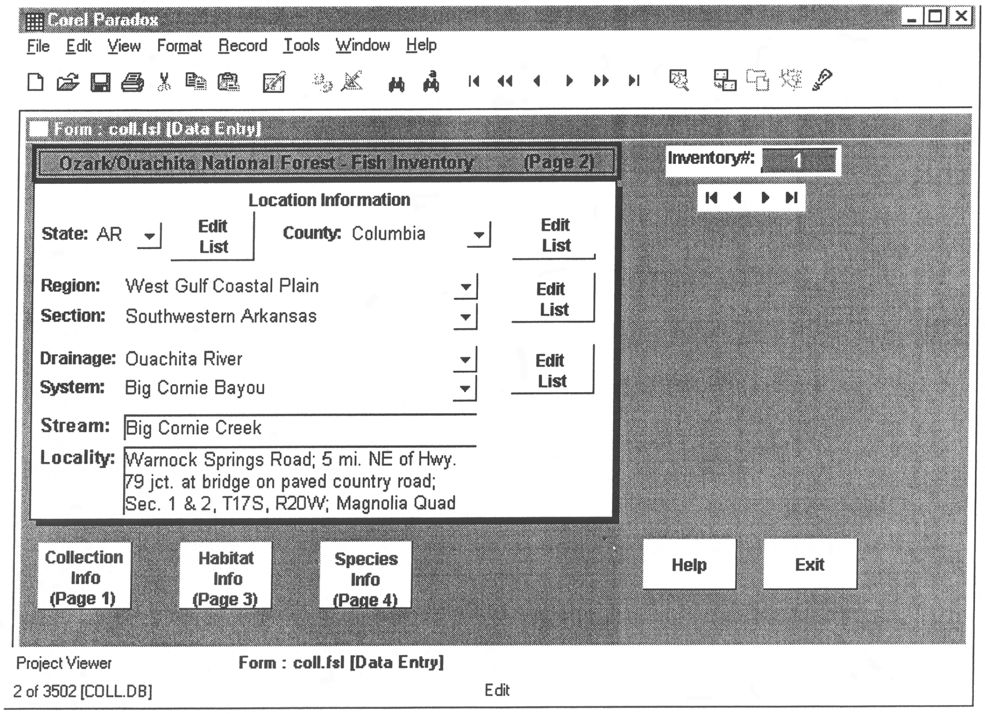
DATABASE DESIGN

The initial design goal was to create a database structure that could incorporate diverse sources of data on fishes into a standardized central, digital repository and that would be flexible and extensible enough to meet anticipated future needs. Additional design considerations included standardization, portability, integration with GIS, ease of use, available PC platforms, and support. These factors together with the one-to-many nature of the primary data dictated the need for a relational database system (e.g., for each fish collection, many species were sampled). A relational system provided the needed master-detail database structures. Importantly, a relational database system also provided programming tools for developing modular code units for data entry, queries, and reports, user-friendly graphical user-interfaces (GUI), data entry validation procedures, and on-screen help displays.

Standardization of the fish collection data was a first step in the design process. The data originated from multiple sources with varying sampling techniques, measurement units, and recording methods. Agreement was reached on standardized values or value ranges for most fields. Attributes of standardized fields were stored in separate lookup tables and "related" to the main database tables as needed via a shared code. This approach provided standard variables, and standard categories for development of queries and analyses and reduced data redundancy and table sizes. Input errors were also reduced since users select values from a pick list rather than typing in the entry. In addition, comment fields were built into the design to capture field observations or specimen conditions that were important but not amenable to standardization.

Due to the magnitude of information related to each collection, data fields were grouped into four major categories, each with a separate data entry screen:

1. Location information (fig. 1), including county; physiographic region and section; river drainage and system; stream name; and exact locality of the sampling site



Form : coll.fsl [Data Entry]

Ozark/Ouachita National Forest - Fish Inventory (Page 2)

Inventory#: 1

Location Information

State: AR

County: Columbia

Region: West Gulf Coastal Plain

Section: Southwestern Arkansas

Drainage: Ouachita River

System: Big Cornie Bayou

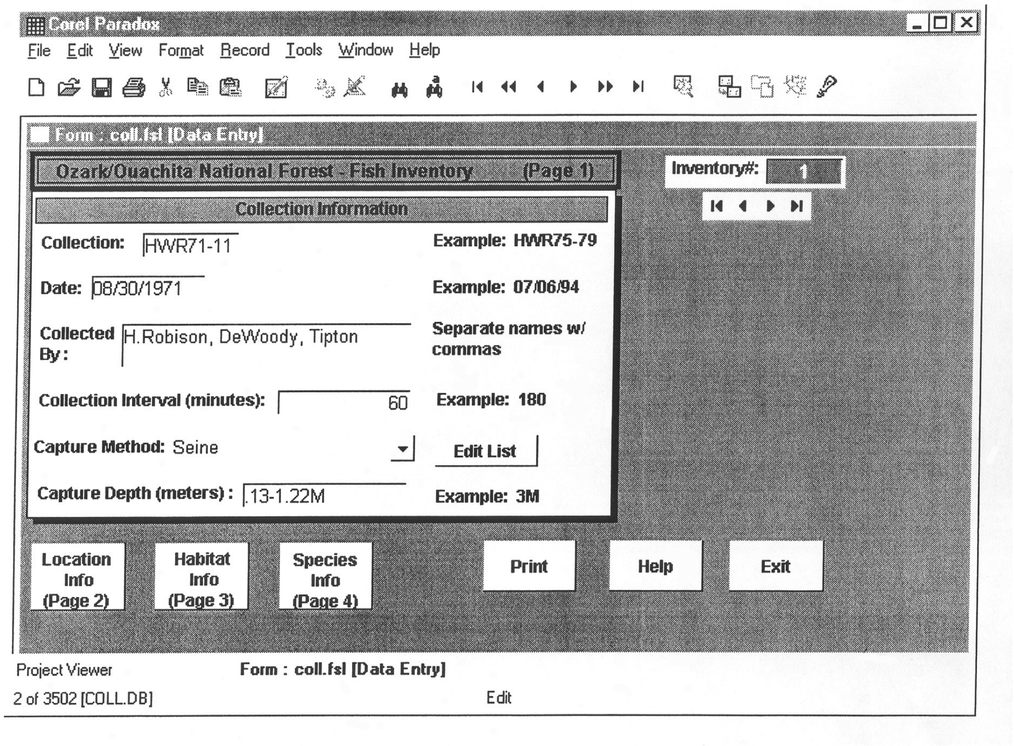
Stream: Big Cornie Creek

Locality: Warnock Springs Road; 5 mi. NE of Hwy. 79 jct. at bridge on paved country road; Sec. 1 & 2, T17S, R20W; Magnolia Quad

2 of 3502 [COLL.DB]

Figure 1—Data entry screen for location information.

2. Collection information (fig. 2), including collection or field identification number; collection date; collection interval, depth, and methods; and collectors
3. Habitat information (fig. 3), including 11 environmental variables: turbidity; vegetation; substrate; geology; shoreline condition; percent riffle; water depth and width; current; stream width; and water and air temperatures
4. Species information (fig. 4), including scientific name and family; number of individual species sampled; museum number; total length, standard length, and notes concerning sex, weight, spawning colors, and spawning condition.



Ozark/Ouachita National Forest - Fish Inventory (Page 1)

Inventory#: 1

Collection Information

Collection: HWR71-11 Example: HWR75-79

Date: 08/30/1971 Example: 07/06/94

Collected By: H.Robison, DeWoody, Tipton Separate names w/ commas

Collection Interval (minutes): 60 Example: 180

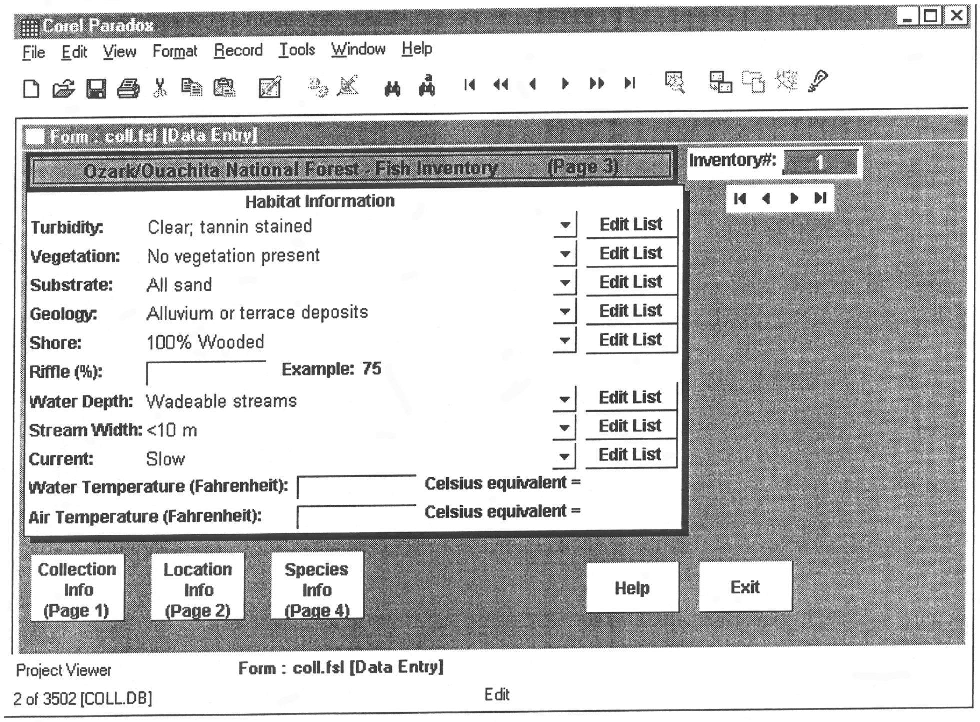
Capture Method: Seine

Capture Depth (meters): .13-1.22M Example: 3M

Form : coll.fsl [Data Entry]

2 of 3502 [COLL.DB]

Figure 2—Data entry screen for collection information.



Ozark/Ouachita National Forest - Fish Inventory (Page 3)

Inventory#: 1

Habitat Information

Turbidity: Clear; tannin stained

Vegetation: No vegetation present

Substrate: All sand

Geology: Alluvium or terrace deposits

Shore: 100% Wooded

Riffle (%): Example: 75

Water Depth: Wadeable streams

Stream Width: <10 m

Current: Slow

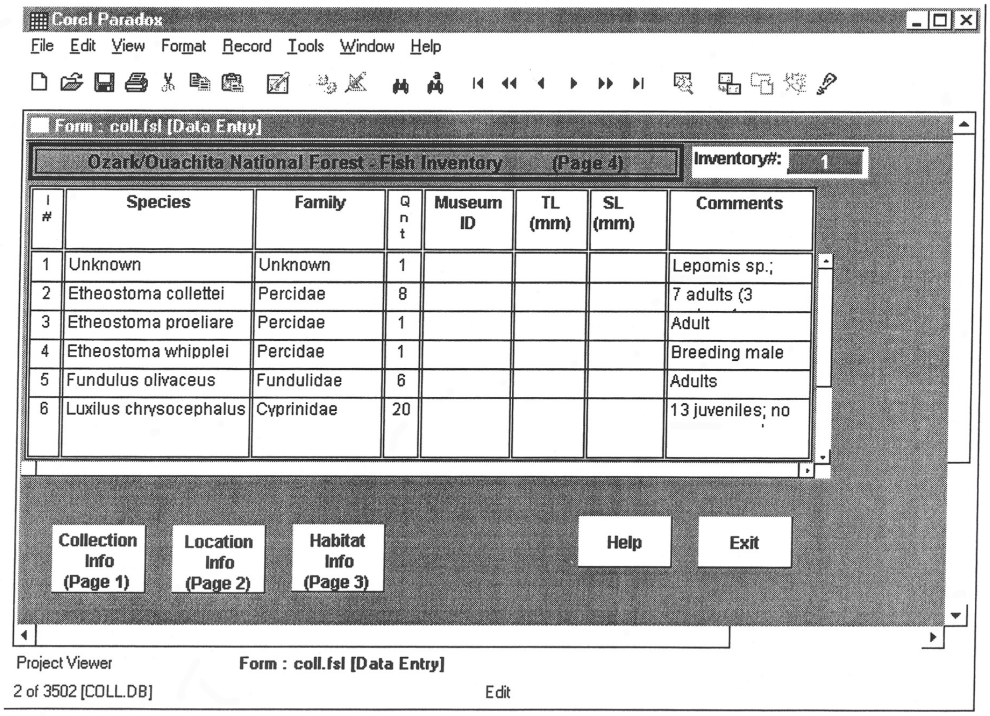
Water Temperature (Fahrenheit): Celsius equivalent =

Air Temperature (Fahrenheit): Celsius equivalent =

Form : coll.fsl [Data Entry]

2 of 3502 [COLL.DB]

Figure 3—Data entry screen for habitat information.



Ozark/Ouachita National Forest - Fish Inventory (Page 4)

Inventory#: 1

I #	Species	Family	Qnt	Museum ID	TL (mm)	SL (mm)	Comments
1	Unknown	Unknown	1				Lepomis sp.;
2	Etheostoma collettei	Percidae	8				7 adults (3
3	Etheostoma proeliare	Percidae	1				Adult
4	Etheostoma whipplei	Percidae	1				Breeding male
5	Fundulus olivaceus	Fundulidae	6				Adults
6	Luxilus chrysocephalus	Cyprinidae	20				13 juveniles; no

Figure 4—Data entry screen for species information.

To enhance accuracy and ensure standardization, the user extracts most field entries from pull-down tables of master lists. Scientific names and respective families for fish species are selected as single entries from a standardized list of all fishes known to occur in Arkansas as documented by Robison and Buchanan (1988) and updated to reflect subsequent taxonomic changes (table 1). A hierarchical list of river drainages and stream systems was adopted from drainage units defined by (Matthews and Robison 1988) (table 2). Similar master lists provide standard political subdivisions (Arkansas county names) and physical and geologic divisions of the state (physiographic region and section, surface geology), (table 3). Others indicate the capture method or provide descriptors of conditions at the collection site and include: air and water temperature, current category, percent riffle habitat, water depth, substrate type, turbidity category, stream size, shore classes, and types of aquatic vegetation (table 4). As needed the master lists may be updated or expanded to accommodate other categories of information or create additional entry fields.

Table 1—Sample from species master list

Species	Family	Species code
Acipenser fulvescens	Acipenseridae	1
Alosa alabamae	Clupeidae	2
Alosa chrysochloris	Clupeidae	3
Ambloplites ariommus	Centrarchidae	4
Ambloplites constellatus	Centrarchidae	5
Ambloplites rupestris	Centrarchidae	6
Amblyopsis rosae	Amblyopsidae	7
Ameiurus catus	Ictaluridae	8
Ameiurus melas	Ictaluridae	9
Ameiurus natalis	Ictaluridae	10
Ameiurus nebulosus	Ictaluridae	11
Amia calva	Amiidae	12
Ammocrypta clara	Percidae	13
Ammocrypta vivax	Percidae	14
Anguilla rostrata	Anguillidae	15

Table 2—Sample from system-drainage master list

Code	System	Drainage
0	0-Unknown	0-Unknown
1	Mississippi River-Blytheville	Mississippi River
2	Mississippi River-West Memphis	Mississippi River
3	Mississippi River-Helena	Mississippi River
4	Mississippi River-Eudora	Mississippi River
5	Upper St. Francis River	St. Francis River
6	St. Francis River	St. Francis River
7	Tyronza River	St. Francis River
8	L'Anguille River	St. Francis River
9	Fifteen Mile Bayou	St. Francis River
10	Upper White River	White River
11	War Eagle Creek	White River
12	Kings River	White River
13	Long Terrapin-Dry Creek	White River
14	Bull Shoals-White River	White River
15	Crooked Creek	White River

Table 3—Physiographic section (region) and geology master lists

Section (Region)	Geology
Arkansas Valley (Arkansas Valley)	Alluvium or terrace deposits
Athens Piedmont Plateau (Ouachita Mtns.)	Chert
Boston Mtns. (Ozark Mtns.)	Dolomite
Bottomlands (MS Alluvial Plain)	Limestone
Central Ouachita Mtns. (Ouachita Mtns.)	Sandstone
Crowley’s Ridge (Crowley’s Ridge)	Shale
Fourche Mtns. (Ouachita Mtns.)	Unknown
Loessial Plains (MS Alluvial Plain)	
Salem Plateau (Ozark Mtns.)	
Southcentral Arkansas (West Gulf Coastal Plain)	
Southwestern Arkansas (West Gulf Coastal Plain)	
Springfield Plateau (Ozark Mtns.)	
Unknown (Unknown)	

Table 4—Master list of valid environmental categories

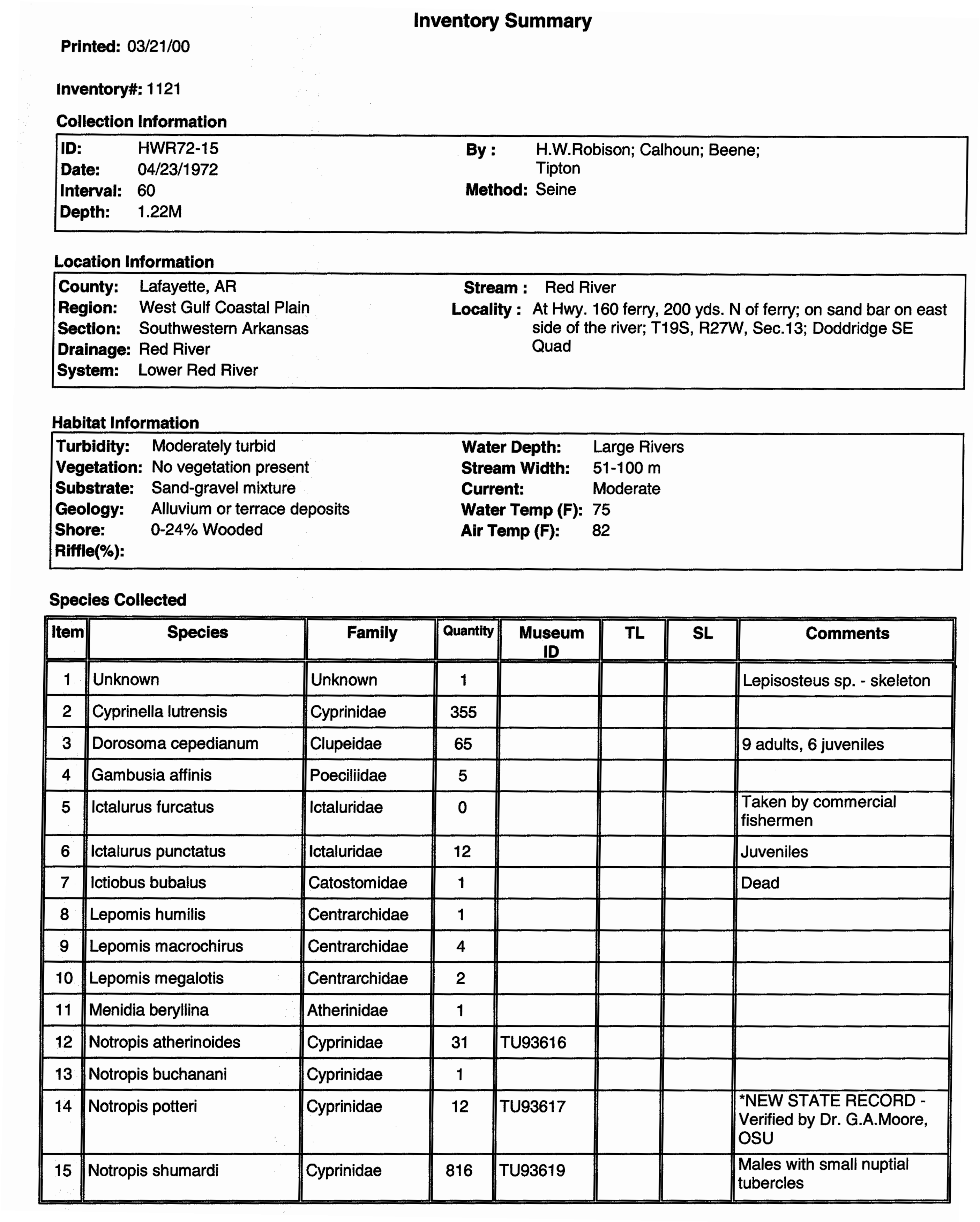
Environmental variables and values
Stream_width:
10-25 m
26-50 m
51-100 m
<10 m
>100 m
Unknown
Water_depth:
Large rivers
Medium-sized rivers
Streams w/pools too deep to wade
Tiniest headwater streams
Unknown
Wadeable streams
Current:
Moderate
None
Slow
Swift
Unknown
Turbidity:
Clear; distinct greenish cast
Clear; tannin stained
Moderately turbid
Slightly turbid
Spring-fed; very clear
Strongly turbid
Unknown
Substrate:
All gravel (mostly small)
All sand
Boulder and bedrock
Gravel-rubble mixture
Mud-sand mixture
Sand-gravel mixture
Unknown
Shore:
0-24 percent wooded
100 percent wooded
25-49 percent wooded
50-74 percent wooded
75-99 percent wooded
Unknown
Vegetation:
Aquatic vegetation beds of rooted aquatic plants
Justicia at stream margins
No vegetation present
Substrate covered with algae
Unknown

Help screens were used to clarify instructions and provide examples of valid entries. Techniques for field-entry validation (e.g., numeric range checking) were employed as appropriate, and printable summary reports (fig. 5) aided verifying entry accuracy. An export routine for creating delimited-text files (ASCII) was provided to ensure the database could later be moved to other systems. The export routine also served as a secondary backup mechanism.

SOFTWARE EVOLUTION AND SYSTEM REQUIREMENTS

The custom application, ONF FISH, has undergone a series of revisions since its inception. The database was originally developed for the DOS platform using Paradox, a relational database development product of Borland (now Inprise). Factors in the selection of Paradox over other relational database systems included PC system requirements (Paradox requires less resources than other products, such as PC Oracle), Borland’s reputation in database development and programming arenas, and previous developer experience with Paradox. The application includes four basic modules accessible from a single, menu-driven, graphical interface: Data, Reports, Queries, and Utilities. Emphasis to-date has been on data entry and editing, but each module can easily be further expanded as user needs evolve.

The last major revision converted the application to object-based Paradox for Windows 5.0 and the Windows 3.1 platforms. The application now functions under Windows ’95 and Paradox for Windows 7.0 (a Corel product). It is currently being ported to Delphi 5.0, a Windows ’9x object-oriented development package originally developed by Borland and now owned by Corel as a result of their recent acquisition of Inprise. The new application will have the look-and-feel of a Windows ’95 or Windows ’98 application. Although still based on Paradox tables, the new application will not require Paradox as an underlying package at runtime. It will allow the use of tables created in any relational database system



Inventory Summary

Printed: 03/21/00

Inventory#: 1121

Collection Information

ID:	HWR72-15	**By :**	H.W.Robison; Calhoun; Beene; Tipton
Date:	04/23/1972	**Method:**	Seine
Interval:	60		
Depth:	1.22M		

Location Information

County:	Lafayette, AR	**Stream :**	Red River
Region:	West Gulf Coastal Plain	**Locality :**	At Hwy. 160 ferry, 200 yds. N of ferry; on sand bar on east side of the river; T19S, R27W, Sec.13; Doddridge SE Quad
Section:	Southwestern Arkansas		
Drainage:	Red River		
System:	Lower Red River		

Habitat Information

Turbidity:	Moderately turbid	**Water Depth:**	Large Rivers
Vegetation:	No vegetation present	**Stream Width:**	51-100 m
Substrate:	Sand-gravel mixture	**Current:**	Moderate
Geology:	Alluvium or terrace deposits	**Water Temp (F):**	75
Shore:	0-24% Wooded	**Air Temp (F):**	82
Riffle(%):			

Species Collected

Item	Species	Family	Quantity	Museum ID	TL	SL	Comments
1	Unknown	Unknown	1				Lepisosteus sp. - skeleton
2	Cyprinella lutrensis	Cyprinidae	355				
3	Dorosoma cepedianum	Clupeidae	65				9 adults, 6 juveniles
4	Gambusia affinis	Poeciliidae	5				
5	Ictalurus furcatus	Ictaluridae	0				Taken by commercial fishermen
6	Ictalurus punctatus	Ictaluridae	12				Juveniles
7	Ictiobus bubalus	Catostomidae	1				Dead
8	Lepomis humilis	Centrarchidae	1				
9	Lepomis macrochirus	Centrarchidae	4				
10	Lepomis megalotis	Centrarchidae	2				
11	Menidia beryllina	Atherinidae	1				
12	Notropis atherinoides	Cyprinidae	31	TU93616			
13	Notropis buchanani	Cyprinidae	1				
14	Notropis potteri	Cyprinidae	12	TU93617			*NEW STATE RECORD - Verified by Dr. G.A.Moore, OSU
15	Notropis shumardi	Cyprinidae	816	TU93619			Males with small nuptial tubercles

Figure 5—Sample inventory report.

for which an Open Database Connectivity (ODBC) driver is available (including Paradox, Access, Oracle, or others). The new version will take advantage of the newest enhancements of Delphi, one of the most powerful Rapid Application Development (RAD) packages available today.

The current version requires at least an 80386-based PC with Windows ’95, Paradox for Windows 7.0, 32MB RAM, and approximately 40MB free disk space for the ONF FISH application. However, a Pentium or better system with 64MB RAM is strongly recommended and will be required for future Delphi-based versions.

INTEGRATION WITH GEOGRAPHIC INFORMATION SYSTEM

In conjunction with development of the fish database application, efforts were made to ensure the information could be easily integrated into a GIS. As site-specific collections were entered, they were also located on paper copies of USGS 7.5 minute topographic maps (1:24,000) and coded with a unique identification number. Those maps are being maintained and continually updated by the first author as a physical record of fish collection localities in Arkansas. In late 1999, in an effort led by Alan Clingenpeel (ONF) and Brian Wagner (AGFC), all sampling locations were georeferenced to state-wide coverages using PC ArcView, a product of Environmental Software Research Institute. Each sampling point on the paper maps was matched to the same point on a 1:24,000 digital topographic coverage of Arkansas. The resulting point coverage uses an ArcView table linking collection identification numbers with point labels. This table will be merged into the original fish database application tables so that all the fish data attributes may be used in map creation and geo-spatial analysis.

CONCLUSION

A digital, database repository linked to GIS is now developed for over 3,500 collections of fishes covering all of Arkansas. Importantly, the foundation of the database, the individual collection records, were critically examined so that the information is up-to-date and as error free as possible (Robison and Buchanan 1998 and subsequent updates by Robison). The abundance and distribution of the fishes of Arkansas are linked intimately to the habitat and water quality condition of streams and rivers these animals inhabit (Matthews and Robison 1988, 1998; Matthews and others 1992). As such, the database of fishes will be extremely useful to natural resource agencies in management, planning, and monitoring. The database gives natural resource managers an enhanced ability to examine fish distribution in association with rehabilitation, enhancement, or remediation of the state’s running waters.

Future applications of the database are limited primarily by one’s vision. The fish data application is currently extensible and will be even more so after full conversion to Delphi, which supports technologies such as Open Database Connectivity (ODBC), Object Linking and Embedding (OLE), and Active Data Objects (ADO) which would allow widespread sharing of data with other applications. Uses may ultimately extend well beyond management. Both Delphi and certain GIS products could be used to extend database products to the web. For example, web-based interactive maps could be created that display recent versus historical fish distributions. Guides to identification of fish species of Arkansas could be developed complete with photographs and detailed distributional maps for each species. Integration of the database with GIS provides managers with decision-making tools and visual communication modes that assist in prioritizing allocation of scarce resource management dollars, open the door to thoroughly examining management alternatives, and help convey and justify management decisions. In sum, the database is a powerful natural resource tool for the USDA Forest Service, The Nature Conservancy, the AGFC, the Department of Arkansas Natural Heritage, and state, federal, and local entities in future planning, monitoring, and management efforts.

ACKNOWLEDGMENTS

The assistance of a number of government agencies and individuals who aided in the realization of a long-held dream of Robison to computerize the fish collection records of Arkansas is most graciously appreciated. First, thanks go to the USDA Forest Service, primarily the Ouachita National Forest and Ecosystem Management Large-Scale Research Program, Southern Research Station (both in Hot Springs, AR) and the Center for Bottomland Hardwoods Research, Southern Research Station (Oxford, MS) and also The Nature Conservancy, for supplying initial and continued funding in the early phases of this project. Especially important in recognizing the value and supporting the project were Alan Clingenpeel, Jim Guldin, William Pell, and Richard Standage (USDA Forest Service), and Douglas Zollner of The Nature Conservancy. We also wish to thank the Arkansas Game and Fish Commission, which in 1998-1999 funded the addition of Arkansas fish records outside of the National Forest. Brian Wagner of the Non-Game Section, Arkansas Game and Fish Commission, was instrumental in supporting the effort.

Research support was provided through the “Ecosystem Management Large-Scale Research Program” dated 1994 through September 1995 under Challenge Cost-Share (CCS) agreement. Special thanks and appreciation is expressed to Thomas M. Buchanan (Westark College, AR), for his willingness to share important data on Arkansas fish collections that he has made throughout the State. Other ichthyologists who graciously shared field notes on fish collections include: John L. Harris (Arkansas Highway and Transportation Department), George L. Harp (Arkansas State University), William J. Matthews and Edie Marsh-Matthews (University of Oklahoma), and Betty Crump (Caddo Ranger District, ONF).

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Session 9

Other Ecosystem Management Research in the Interior Highlands

Charles Richmond, Moderator

GYPSY MOTH DEFOLIATION POTENTIAL IN THE OUACHITA/OZARK HIGHLANDS REGION

**Andrew M. Liebhold, Kurt W. Gottschalk,
James M. Guldin, and Rose-Marie Muzika**[1]

Abstract—The gypsy moth is expanding its range in North America and is likely to invade the Ouachita/ Ozark Highlands region sometime during this century. A previous analysis indicated that forests in this area are among the most susceptible in North America to defoliation by this insect. We used USDA Forest Service, Forest Inventory and Analysis data to evaluate forest susceptibility in the region. Susceptibility was estimated as the proportion of basal area composed of tree species preferred by gypsy moth caterpillars. Analyses were stratified by ecological land type and land ownership. Forest susceptibility is highest to the north, in the Ozark Highland's area; ca. 80 percent of the forests in this area have high to very high susceptibility to defoliation. Forest susceptibility was lower to the south in the Ouachita region. This trend in susceptibility reflects the increased pine component in southern portions of the region (pine species are not highly preferred by the gypsy moth). South of the Ozark Highlands, in the Boston and Ouachita Mountains, the lower proportion of susceptible forests is lower in land owned by the forest industry, presumably because of more intensive management of softwoods. Most forests in the Ouachita/Ozark region are susceptible to gypsy moth defoliation. Should populations become established in this area, intense defoliation could result in extensive ecological and economic consequences.

INTRODUCTION

Since the gypsy moth was originally introduced near Boston in 1868 or 1869, it has been slowly expanding its range to include the entire Northeastern United States and portions of Virginia, West Virginia, North Carolina, Ohio, and Michigan (Liebhold and others 1992, 1997a). In many of the forested regions where this insect has become established, outbreaks occur intermittently every 5 to 15 years (Williams and Liebhold 1995). During these outbreaks, defoliation of host trees can be extensive and result in severe ecological and economic effects. It is inevitable that the gypsy moth will continue to spread to the south and west during this century.

The Ouachita/Ozark Highlands region is currently ca. 750 km from the expanding front of gypsy moth defoliation. Based on an historical rate of spread of ca. 21 km/yr (Liebhold and others 1992), we might expect the first defoliation in the region around the year 2035, though it is possible that gypsy moth will be introduced accidentally to the area before then. Isolated infestations have been discovered in the region but eradication efforts to date have been successful. Should future eradication efforts fail or if this strategy is abandoned, defoliating populations likely will appear before 2035. The USDA Forest Service has initiated a program aimed at slowing the spread of the gypsy moth in the Midwest and elsewhere (Leonard and Sharov 1995, Sharov and others 1998). If this program is continued and successful, defoliating populations may not invade the region until well after 2035.

To plan for the management of the gypsy moth, the distribution of susceptible stands must be limited in currently uninfested areas. Liebhold and others (1997a, 1997b) analyzed forest inventory data from across the conterminous United States to evaluate the susceptibility of all forests to gypsy moth defoliation. The analysis indicated that the Ouachita/Ozark Highland's had one of the highest concentrations of forests that are highly susceptible to the gypsy moth. In this paper we provide a more detailed description of projected forest susceptibility to gypsy within this region.

METHODS

The gypsy moth is a polyphagous insect; North American populations feed on more than 300 different shrub and tree species (Leonard 1981, Liebhold and others 1995). Despite this breadth of host preference, there is considerable variation among Northeastern North American forests in their susceptibility to defoliation. We define susceptibility as the probability or frequency of defoliation given an established gypsy moth population in the area (Gottschalk 1993).

Several studies that have focused on relating various characteristics of forests to susceptibility have yielded susceptibility models of varying levels of complexity. Perhaps the most important factor affecting stand susceptibility is the proportion of basal area represented by species that are highly preferred by the gypsy moth (Herrick and Gansner 1986). Other variables, such as the predominance of chestnut oak, the abundance of tree structural features; e.g., bark flaps, and various site characteristics; e.g., soils, are correlated with susceptibility (Bess and others 1947, Herrick and Gunner 1986, Valentine and Houston 1979). However because of the dissimilarity between forests of the Northeast where the earlier studies were conducted and Ouachita/ Ozark forests; e.g., chestnut oak does not grow there, it is questionable whether previous correlations could be extrapolated for this region. Also, relatively few site and tree characteristics used in earlier studies had been measured in the forest inventory data available to us. As a result, we excluded all plot characteristics except species composition.

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Table 1—Number of FIA plots used in the analysis

State	Year of inventory	Number of forested plots
Arkansas	1995	2,796
Missouri	1989	6,645
Oklahoma	1993	663

FIA = Forest Inventory and Analysis.

Thus we adopted proportion of basal area represented by preferred species as the measure of forest susceptibility. Montgomery's (1991) three-way classification (susceptible, resistant, immune) was used to categorize each tree species. This classification was based on a summary of field and laboratory studies, as well as extrapolations based on taxonomic affinity. It is described in detail in Liebhold and others (1995). In a previous study in which similar data were used (Liebhold and others 1997b), proportion of basal area was highly correlated with defoliation frequency at the county level for States located within the infested area.

Assessment of forest susceptibility was based on USDA Forest Service, Forest Inventory and Analysis (FIA) data compiled in the Eastside Forest Data Base (Hansen and others 1993). These data are collected in statewide inventories conducted every 10 years. Inventory data are collected at permanent plots located throughout forested areas of each State. We selected data from the 9,777 plots in Arkansas, Missouri, and Oklahoma that fell within the Ouachita/Ozark Highlands (table 1) (fig. 1).

Summaries of forest susceptibility were stratified using an ecological land type classification developed by Keys and others (1995). Expanded, from an earlier classification by Bailey (1995) the hierarchy of Keys and others begins with domain at the global ecoregion level and can be refined to the landscape and land unit, which can be hundreds to < 10 ha in size. Provinces are regional ecological delineations. The areas of interest for this study in the Ozark-Ouachita area fall within four provinces. The Ozark Highlands section is categorized within the Eastern Broadleaf Forest (continental) Province. The Boston Mountains section is the only one within the Ozark Broadleaf Forest—Meadow Province. The Arkansas Valley section is one of seven sections within the Southeastern Mixed Forest Province. The Ouachita Mountains section is the only one in the Ouachita Mixed Forest—Meadow Province. Twenty subsections fall within these four sections (table 2). The location of each FIA plot within a subsection was determined by comparing approximate survey plot locations with the digital map version of the Keys and others (1995) classification using a GIS. When plots fell near the margin of a subsection, final classification into the appropriate subsection was determined by subjectively comparing individual plot locations with a digital elevation model of the region (Foti and Bukenhofer 1999).

All inventory data contained information about individual trees and plots. Individual tree records were used to sum total basal area by species for each plot. These plot records were then expanded (using appropriate factors) to subsection-level estimates of basal area per acre. This information was then used to estimate the proportion of forested land in each subsection that fell into one of four susceptibility classes based on the percentage of basal area composed of preferred tree species: 0 to 20 percent = low, 20 to 50 percent = moderate, 50 to 80 percent = high, 80 to 100 percent = very high. We used this susceptibility classification scheme in our earlier analysis (Liebhold and others 1997b) because historical patterns of defoliation seem to be closely related to these classes (Gottschalk 1993). The type of land owner is recorded at all FIA plot locations, so we estimated the proportion of land falling in each of the four susceptibility classes for public forest, forest industry, and other private land for each subsection.

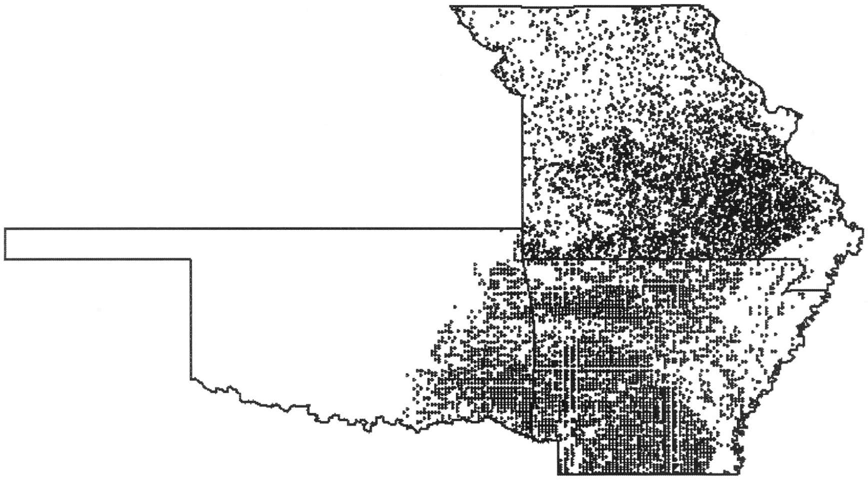

Figure 1—Location of USDA Forest Service, Forest Inventory and Analysis plots in Missouri, Arkansas, and Oklahoma.

Table 2—Characteristics of ecological landtype subsections in the Ouachita and Ozark Highland's region[a]

Section	Subsection	Number of plots	Preferred species
			% BA
Ozark Highlands (222A)			
St. Francis Knobs and Basins	222Aa	256	64
Central Plateau	222Ab	921	70
Osage River Hills	222Ac	279	75
Gasconade River Hills	222Ad	211	66
Meramac River Hills	222Ae	301	71
Current River Hills	222Af	417	66
White River Hills	222Ag	605	59
Elk River Hills	222Ah	87	74
Black River Ozark Border	222Al	210	67
Springfield Plain	222Am	215	65
Springfield Plateau	222An	225	65
Boston Mountains (M222A)			
Boston Mountains	M222AA	157	55
Boston Hills	M222AB	374	51
Arkansas Valley (231G)			
Eastern Arkansas Valley and Ridges	231GA	127	39
Mount Magazine	231GB	115	40
Western Arkansas Valley and Ridges	231GC	127	51
Ouachita Mountains (M231A)			
Fourche Mountains	M231AA	372	38
West Central Ouachita Mountains	M231AB	239	34
East Central Ouachita Mountains	M231AC	232	38
Athens Piedmont Plateau	M231AD	132	23

BA = basal area.

[a] Subsection codes in parentheses.

RESULTS AND DISCUSSION

The Ouachita/Ozark Highlands region is a hilly to mountainous area that differs from the surrounding area both in topography and geological origin. The Highlands consist of two ecological sections—the Ozark Mountains in southern Missouri, northern Arkansas, and northeastern Oklahoma; and the Ouachita Mountains in western Arkansas and eastern Oklahoma. Table 2 shows the estimated percentage of basal area composed of tree species preferred by the gypsy moth in each of these sections. As stated earlier, we used this percentage as our metric of forest susceptibility to the gypsy moth. The Ozark Highlands section (222A) generally had the most susceptible mixtures of species of the four sections. These same data were mapped graphically (fig. 2) and clearly show a trend with increasing proportions of susceptible species, particularly oaks, in the northern portion of the region. Examination at a coarser scale of forest type reveals a north-south trend in forest type that reinforces our analysis (fig. 3).

The dominant forest type group in the northern portion of the Ouachita/Ozark Highlands is oak-hickory while the loblolly-shortleaf pine type group dominates in the southern portion of the region. Each of the forest type groups in figure 3 represents many different specific forest types (Eyre 1980). Nevertheless, these maps depict a real trend in the region; that is increased dominance of oak to the north and increased dominance of pine to the south. Analyzing FIA data from the Ouachita/Ozark Highland regions Guldin and others (in press) found that the proportion of stands in the oak-hickory forest type group was greater in the Ozark Mountain section (north) and that the proportion of stands in the loblolly-shortleaf type group was greatest in the Ouachita Mountains section.

The observed trend from oak-dominated to pine-dominated forests from north to south in the Ouachita/Ozark regions reflect in part an historical anomaly. At one time much of the forested area of the Missouri Ozark Mountains currently dominated by oak-hickory forests were dominated by pure shortleaf pine or mixed pine-oak forests. Before 1880, pine and oak-pine cover types were estimated at 6.6 million acres in Missouri (Liming 1946). Extensive logging and changes in fire frequency have resulted in shifts in species composition (Batek and others 1999).

Our results were derived from summaries pooled across all inventory plots in land type subsections. The question arises: how is susceptibility to gypsy moth distributed among individual stands? Figure 4 shows estimates of the percentages

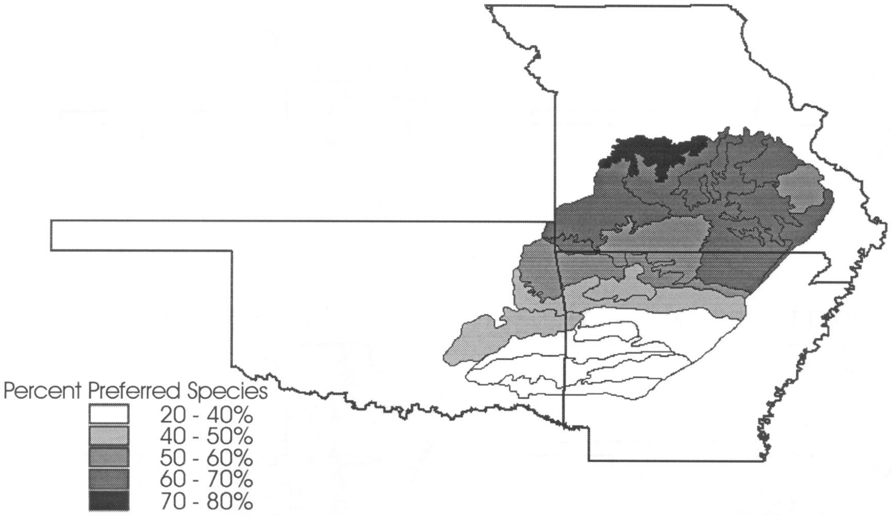

Figure 2—Map of the Ouachita and Ozark Highlands region showing the proportion of basal area composed of tree species susceptible to the gypsy moth by ecological subsection.

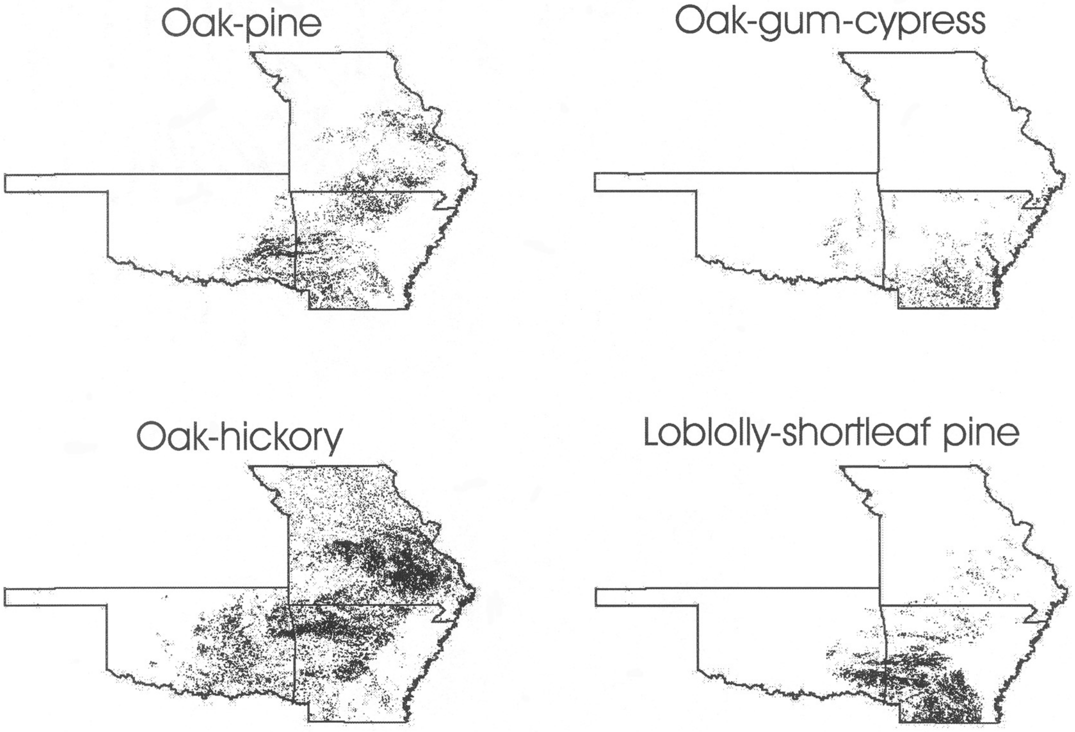

Figure 3—Distribution of four major forest types in the Ouachita and Ozark Highlands region. Maps were modified from maps described in Zhu and Evans (1992).

of land area covered by forest of varying levels of susceptibility. These data indicate that the Ozark Highlands had the greatest area in highly susceptible stands and the Ouachita Mountains section had the highest area in stands with low susceptibility. This reflects the same latitudinal trend seen in figure 2 and presumably is due to the higher pine component in the more southerly subsections.

Separate analysis of susceptibility by ownership in the Boston and Ouachita Mountains revealed a lower proportion of susceptible forests on forest industry lands (fig. 5). Much of the industry land in these sections is managed for softwood production. This higher pine component would explain the lower susceptibility in these sections compared to public and other private lands. Guldin and others (in press), by contrast, forest industry land in the Ozark Highlands is largely

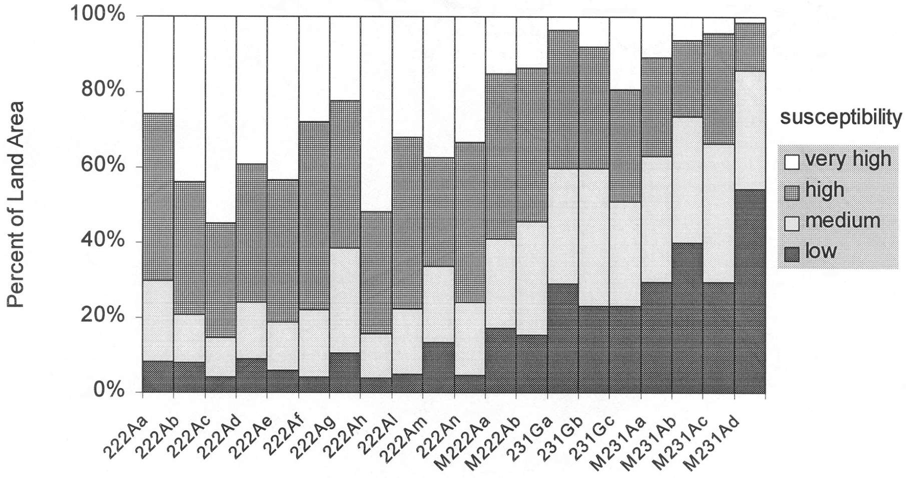

Figure 4—Percentages of forest land area falling into each of four gypsy moth susceptibility classes for each subsection. Susceptibility classes were based on percent basal area composed of species preferred by the gypsy moth: low = 0 to 20 percent, medium = 20 to 50 percent, high = 50 to 80 percent, and very high 80 to 100 percent.

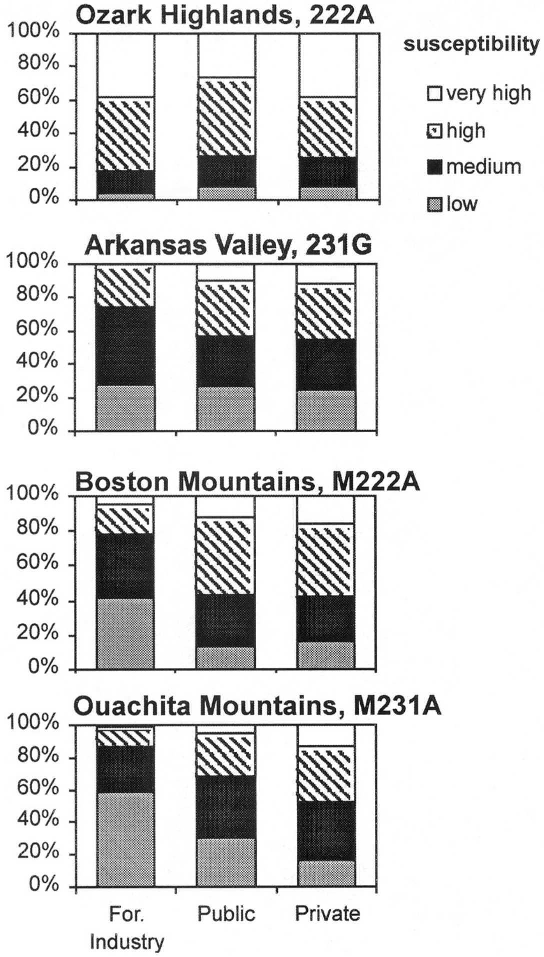

Figure 5—Percentages of forest land area falling into four gypsy moth susceptibility classes for different land ownership classes in each land type section. Susceptibility classes were based on percent basal area composed of species preferred by the gypsy moth: low = 0 to 20 percent, medium = 20 to 50 percent, high = 50 to 80 percent, and very high 80 to 100 percent.

managed for oak and seems to be at least as susceptible as public and other private land (fig. 5).

The classification of forest susceptibility solely on the basis of species composition may not capture all of the differences with respect to the true susceptibility to gypsy moth defoliation. One trend that can be extracted from studies of gypsy moth susceptibility in the Northeastern United States apart from the association with dominance by preferred species is that defoliation tends to be more intense on poor sites (Herrick and Gansner 1986, Valentine and Houston 1979). Guldin and others (2000) found that site quality generally was greater to the south in the Ouachita region and lower to the north in the Ozark Highlands. If forest susceptibility to the gyspy moth continues to be associated with poor sites, as has occurred in the northeast, this would tend to reinforce the trend observed in figures 3 and 4, that is forest susceptibility is greater in the northern portion of the region.

CONCLUSION

The range of the gypsy moth is likely to continue to expand and defoliating populations are likely to occur in the Ouachita/Ozark Highlands region during this century. An analysis by Liebhold and others (1997a, 1997b) indicated that forests in this area are among the most susceptible in the Nation to defoliation by this insect. The establishment of gypsy moth populations in the region could result in extensive ecological and economic consequences. Our analysis indicates that forest susceptibility is particularly high to the north, in the Ozark Highlands. The data in figure 4 indicate that ca. 80 percent of the forests in this area have high to very high susceptibility to defoliation. It is likely that gypsy moth populations will not become established for another 30 years. Still forest management practices in this area that promote pines or other species not favored by the gypsy moth should reduce the future impact of this insect pest (Gottschalk 1993).

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HERPETOLOGICAL HABITAT RELATIONS IN THE OUACHITA MOUNTAINS, ARKANSAS

Doyle L. Crosswhite, Stanley F. Fox, and Ronald E. Thill[1]

Abstract—We studied habitat relationships of the herpetofauna inhabiting managed pine-oak woodlands of the Ouachita Mountains, Arkansas. We used drift fence arrays with pitfall and double-ended funnel traps to sample two replications each of three treatments: young clearcuts, selectively harvested stands, and late-rotation untreated controls. Our objectives were to compare herpetofaunal communities among these treatments and to quantify habitat relationships. Ninety-one days of trapping over two field seasons yielded 633 captures representing 35 species. Canonical correspondence analysis indicated that species composition differed significantly among treatments. The most distinct separation of species groups was between reptiles and amphibians; reptiles were far more abundant in the young, xeric clearcuts, while amphibians were most abundant in the other two treatments. Four habitat parameters (canopy coverage, litter depth, woody plant cover, and large, woody debris) explained much of the variation in species composition among sample sites. Several species showed clear preferences for particular habitats.

INTRODUCTION

Several geographical and geological factors have contributed to the unique fauna and flora of the Interior Highlands of Arkansas, Oklahoma, and Missouri. Unlike the Southwestern United States, the Interior Highlands were not covered by shallow inland seas during the Cretaceous period (Dowling 1956); consequently, these highlands served as an island refuge. The region also may have served as a refuge for plants and animals during the Pleistocene epoch when glaciers covered adjacent northern regions and the formation of the Arkansas River divided the region into the Ozark Mountains to the north and the Ouachitas to the south (Dowling 1956). During the late Cenozoic era, sediments that had been deposited by inland seas were eroded, further defining boundaries and isolating the uplift.

The Ouachita uplift has unique habitats that supports a rich flora and fauna, including more than a dozen endemic plant species (Mohlenbrock 1993). The herpetofauna is also rich, with high species densities of both reptiles and amphibians (Kiester 1971). Reptile faunal assemblages are more or less representative of adjacent regions and no endemic species are found within the uplift. Frogs and toads, which are relatively mobile, also are not represented by endemic forms. Salamanders, however, are represented by five endemic species, and several endemic subspecies (Connant and Collins 1998).

Many of the species of reptiles and amphibians in the Ouachitas are relatively uncommon and some are considered threatened due to limited distributions or low population densities. Ashton (1976), Black (1977), and Reagan (1974) list the following as rare or threatened: *Amphiuma tridactylum*, three-toed amphiuma, *Ambystoma annulatum*, ringed salamander, *Ambystoma talpoideum*, mole salamander, *Plethodon ouachitae*, Rich Mountain salamander, *Plethodon caddoensis*, Caddo Mountain salamander, *Hyla avivoca*, bird-voiced tree frog, *Cemophora coccinea*, scarlet snake, and *Terrapene ornata*, ornate box turtle.

Two silvicultural systems are employed in the region: even-aged management (e.g., clearcutting) and selective harvesting. With clearcutting, all or most of the trees are harvested from an area such that the “forest influence” is removed from most of the area (Kimmins 1992). A new population of seedlings is then established through natural regeneration or planting such that one dominant age-class of trees is represented. With selective harvesting, individual trees or groups of trees are removed periodically so that the resulting forest eventually contains trees of several distinct age/size classes (Kimmins 1992). Even-aged silviculture employing clearcutting, site preparation, and planting of pines has been the primary method of pine regeneration on southern forests for >30 years. Although young pine plantations provide excellent habitat for many wildlife species adapted to early successional stages (such as deer, rabbits, and quail), clearcutting is generally detrimental to species that require an abundance of snags and cavity trees, hardwoods, hard mast, woody debris, and other forest habitat features (Kimmins 1992, Thill 1990). It has been shown that some reptiles and amphibians require similar mature habitat features; e.g., oak-hickory habitats supported greater numbers of amphibians than nearby managed-pine habitats in South Carolina (Bennett and others 1980). Similarly, Enge and Marion (1986) found that clearcutting and site preparation in Florida had a negative impact on reptile and amphibian numbers and reptile species richness. The decrease in amphibian numbers in heavily disturbed areas was primarily due to reduced reproductive success in certain species, such as *Scaphiopus* spp., *Rana utricularia*, and *Gastrophryne carolinensis*. Low numbers of young-of-the-year were noted in clearcut areas, possibly due to disappearance of standing water before young frogs and toads could metamorphose. In another study, presence and numbers of amphibians in managed stands were strongly affected by the occurrence and longevity of intermittent ponds and streams during winter (Whiting and others 1987).

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By altering soil structure, hydrology, and horizontal and vertical vegetation structure, clearcutting can affect microclimatic conditions (Geiger 1971), which influences amphibian and reptile distribution and abundance (Ash 1988, Matlack 1994, Pechman and others 1991, Pough and others 1987). These changes result from canopy removal, reduction in moisture-retaining litter, and soil compaction (Bratton 1994, Bury 1983, Raymond and Hardy 1991).

Many terrestrial salamanders require moisture-containing deciduous leaf litter for site colonization (Jaeger 1971). Thus, pure stands of conifers are generally unsuitable for salamanders in the Eastern and Central United States (Bennett and others 1980, Pough and others 1987, Williams and Mullin 1987). In loblolly-shortleaf pine (*Pinus taeda* and *P. echinata*) stands of east Texas, Whiting and others (1987) found that understory development and degree of deciduous litter accumulation strongly influenced herpetofaunal communities.

Petranka and others (1993) compared 5-year-old clearcuts with mature stands over 80 years old and found that terrestrial salamanders were reduced by 75 to 100 percent following clearcutting. Furthermore, Petranka and others (1994) estimated that it would require a century or more for salamander populations to return to predisturbance levels. There is concern that this reduction could produce population bottlenecks and decreased genetic diversity. In some cases, local populations of sedentary species may be prone to extinction (Petranka and others 1993).

On a regional scale, survival of a reduced population depends upon recolonization through immigration from undisturbed areas (Fahrig and Merriam 1994). However, several factors limit salamander immigration: (1) salamanders generally only migrate under a narrow set of environmental conditions, (2) migrating individuals may encounter interspecific competition with other herps, and (3) adult salamanders are often highly philopatric (Petranka 1994, Petranka and others 1993). Consequently, recolonization of heavily disturbed areas by salamanders is slow.

Like amphibians, reptile species richness and community composition are affected by silvicultural treatments (Enge and Marion 1986, Whiting and others 1987). Populations of some reptiles increase in response to clearcutting due to increased prey abundance, creation of favorable microhabitats or refugia, and other factors (Enge and Marrion 1986). *Cnemidophorus sexlineatus*, a cursorial lizard that prefers open sandy areas, increased in abundance following clearcutting (Enge and Marion 1986). Several grassland species were also common in young plantations, including *Thamnophis proximus*, *Masticophis flagellum*, *Lampropeltis calligaster*, and *L. getula* (Whiting and others 1987). Clearcutting typically results in increased small mammal densities and species diversity (Atkinson and Johnson 1979; Kirkland 1977, 1990), providing more prey for snakes that feed primarily on small rodents.

Reptile community composition is related to understory and overstory development as well as presence of woody debris, rocky outcroppings, and prey abundance. Many habitat characteristics affecting herpetofaunal community composition are ultimately dependent upon age of the forest and degree of disturbance.

Because amphibians are often habitat specialists with restricted distributions, they may be valuable indicators of ecosystem health and stability. Despite new evidence that reptiles and amphibians are important components in many ecosystems, they continue to be neglected by land managers (Pough and others 1987). Some management plans may even promote midsuccessional stages to maximize alpha diversity of other taxa at the expense of sensitive reptile and amphibian species (Faaborg 1980, Samson and Knopf 1982).

Our objectives were to (1) determine if herpetofaunal community structure differs among silvicultural treatments in the Ouachitas, (2) quantify microhabitat differences among treatments, and (3) relate herpetological community composition to microhabitat conditions.

MATERIALS AND METHODS

We sampled six stands (two replicates of three treatments) located within Perry County, AR, about 70 km north of Hot Springs. The treatments were young (3 and 5 years old at study initiation) clearcut plantations, late-rotation (80+ years old) naturally regenerated stands (hereafter controls), and selectively harvested stands (table 1). Stands of the first two treatments were managed by the USDA Forest Service; selectively harvested stands were managed by forest industry.

All stands had a predominately south, southeast, or southwest aspect and slopes of 5 to 20 percent. We chose stands with southerly aspect because these best represent sites that the USDA Forest Service manages for *P. echinata* in the Ouachitas.

Table 1—Treatment histories for six study areas in the Ouachita Mountains

Treatment	Year of harvest	Year of burn[a]	Year of herbicide treatment
Selectively harvested	1972,92	1985,88	1973 (2,4,5-T)
Selectively harvested	1976,91	1988	1973 (2,4,5-T)
Control	1912	—	—
Control	1912	—	—
Clearcut	1990	—	1990 (Garlon 3A)
Clearcut	1988	—	1988 (Garlon 3A)

[a] Prescribed burning of understory was conducted to reduce woody debris and young hardwoods.

We established three drift fence arrays with associated pitfall and funnel traps on each site (fig. 1a). The array design was modified from Campbell and Christman (1982), Vogt and Hine (1982), and Jones (1986). Within each stand, arrays were positioned at 100-m intervals along a central transect situated approximately 100 m from roads, streams, or stand borders to minimize treatment confoundments. Each array consisted of three 15.2-m by 30.5-cm wings (galvanized flashing) originating from the center and radiating outward at approximately 120° angles. Drift fences were buried 5 cm in the ground to help prevent burrowing under the fence. An 18.9-l pitfall trap (plastic bucket) was buried at the center of the array and at the end of each of the three wings (fig. 1a). Pitfalls were buried flush with the ground, allowing the drift fence to overhang the lip of the pitfall (fig. 1b). Drain holes were punched in the bottom of each pitfall. Two double-ended funnel traps, one of hardware cloth (0.64 cm mesh) (Fitch 1951) and one of aluminum window screen (Jones 1986), were placed on both sides of each wing for a total of 12 funnel traps per array. Funnel traps were molded and positioned as close to the fence as possible to prevent animals from moving between the funnel traps and the drift fence.

Arrays were installed during March 1993, about 2 months prior to initial trapping. Arrays were checked on alternate days for a total of 91 days during six periods: three in 1993 (May 22 to June 6, June 15 to 30, and July 15 to 25) and three in 1994 (March 6 to 21, May 14 to 29, and June 15 to 29). When pitfalls were not in use, they were closed with

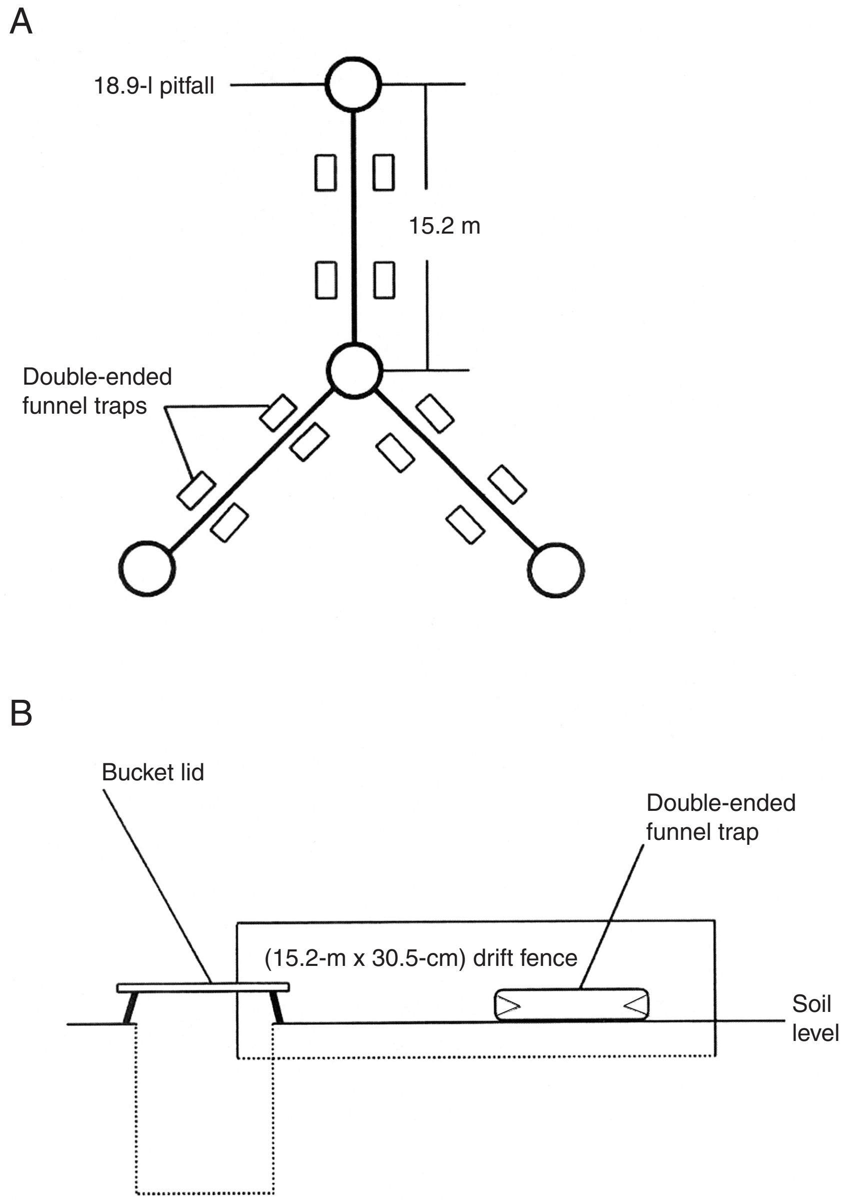

Figure 1—(A) Array design showing configuration of drift fences, pitfall traps, and double-ended funnel traps; (B) side view of an array segment showing the intersection of a pitfall trap with the drift fence.

tight-fitting snap lids; funnel traps were closed by lodging a plug of aluminum foil in their entrances. To minimize mortality from overheating and desiccation, a piece of asphalt shingle was draped over each funnel trap for shade. Pitfalls were shaded using small sticks to prop lids 10 to 15 cm above the container (fig. 1b).

Twelve habitat measurements were collected at each of the 18 arrays during July 15-30, 1993. Habitat changes from 1993 to 1994 were negligible, so the measurements made in 1993 characterize the total study period. Leaf litter, exposed rock, woody plant cover, herbaceous cover, and woody debris were quantified by visually estimating, with an ocular tube, the percent of the ground surface covered by each. Percent coverage by woody debris was recorded in two size classes (Maser and others 1979): total woody debris and large debris with a mean diameter >20 cm. Forest canopy cover was estimated using a spherical densiometer (Lemmon 1957). Pine and hardwood basal area was estimated with a 10-factor English prism. Litter depth was measured with a metric ruler and horizontal cover was estimated using a 1-m by 1-m density board at two heights: resting on the ground and centered at 1 m aboveground (MacArthur and MacArthur 1961). For each array, these data were collected at right angles to the drift fence, 2 m to either side of each of the peripheral pitfall traps, for a total of six points per array. Horizontal cover, litter depth, and all percent coverage estimates were recorded at these points. Basal area was estimated from the center of each pitfall trap. The data for each parameter were then averaged to characterize each array.

DATA ANALYSIS

We employed canonical correspondence analysis (CCA) to test for differences in herpetofaunal communities among silvicultural treatments and to identify associations of habitat variables with the treatments and with particular reptiles and amphibians. CCA is a gradient analysis that utilizes aspects of multivariate regression and correspondence analysis to relate species composition of the samples with measured habitat variables. Ordination axes are constrained such that they are linear combinations of the habitat variables and each subsequent axis explains variation in the data set not already explained by previous axes (i.e., axes are orthogonal). Ordination diagrams show the relationships among species abundances, sites, and habitat variables (Taylor and others 1993, Ter Braak and Smilauer 1998).

In CCA ordination diagrams, sites and species are represented by symbols (points) while habitat variables are represented by vectors. The length of a vector symbolizes the importance of the environmental variable while the direction of vectors indicates the degree of correlation among habitat variables and sites, and/or habitat variables and species. Only the positive end of environmental vectors are shown in the CCA diagrams; therefore, one must remain aware of the equally important negative portion of each vector. For each environmental variable shown in the ordination, one can imagine a vector of equal length extending from the center of the figure and in the opposite direction. The closer environmental vectors are to one another the more they are correlated, and the closer these vectors align with an axis the more the nature of that axis is identified. The location of sites relative to environmental vectors indicates the habitat characteristics of the sites, while the position of species points relative to vectors shows the environmental associations of individual species.

Analyses were performed using the program CANOCO with downweighting of rare species (Ter Braak and Smilauer 1998). Each drift fence array was considered a sample site. Species abundances were $\log_{10}$ transformed and environmental data expressed as proportions were transformed to the arcsine of the square root of the value. For purposes of ordination, it was valid to incorporate the total set of variables, but for purposes of hypothesis testing, the number of habitat variables (12) was large relative to the number of samples (18) (Ter Braak and Smilauer 1998). Therefore, before applying the CCA for hypothesis testing, we reduced the number of habitat variables using Principal Components Analysis (PCA) to identify those variables that were redundant or superfluous. From this analysis, we identified four habitat variables (canopy cover, litter depth, woody plant cover, and large, woody debris) that most influenced the herp community structure. Although canopy cover and litter depth are strongly correlated, both were included in the model because of the known importance of a well-developed litter layer to amphibians (Bury 1983, Diller and Wallace 1994).

Monte Carlo permutation tests (Manly 1992) were used to test the overall effects of (1) treatment and (2) the selected habitat variables on species composition. Monte Carlo permutation tests were also used to test the effect of the first CCA axis (CCA1) for each of the analyses.

RESULTS

We captured 633 individuals representing 35 species of reptiles and amphibians (table 2). Of these, 62 percent (395) were lizards (Phrynosomatidae, Teidae, Scincidae), 26 percent (162) were frogs and toads (Microhylidae, Bufonidae, Ranidae), and 10 percent (66) were snakes (Colubridae, Viperidae). Salamanders and turtles (Ambystomatidae, Plethodontidae and Testudinidae) combined represented < 2 percent of all captures and therefore will only be briefly discussed.

Based on CCA using treatment as the only environmental variable, the pattern of species abundances (overall ordination) was nonrandom along CCA1 (Monte Carlo test, $p < 0.01$). In other words, the herpetofaunal communities differed significantly among the three treatments. Using this same analysis with four preselected variables (canopy cover, litter depth, woody plant cover, and large, woody debris), patterns of species abundances among treatments also differed ($P < 0.01$).

In the ordination using all habitat variables (fig. 2), CCA1 was positively correlated with leaf litter, pine basal area, canopy cover, litter depth, and hardwood basal area; CCA 1 was negatively correlated with horizontal cover at 1 m. Over the first three canonical axes, the three silvicultural treatments were well separated (fig. 3), with CCA1 providing the greatest separation. Overlaying the habitat variables (fig. 2), clearcut stands were characterized by dense ground cover including woody and herbaceous vegetation as well as an

Table 2—Amphibians and reptiles captured using drift fence arrays in the Ouchita Mountains of Arkansas, 1993–94[a]

Species	Acronym	Clearcut	Selectively harvested	Control	Total	(%)
Frogs and Toads						
Bufo americanus	BUFAME	5	65	39	109	(17.2)
Gastrophryne carolinensis	GASCAR	2	37	5	44	(7.0)
Rana clamitans	RANCLA	4	0	2	6	(0.9)
Rana catesbeiana	RANCAT	0	0	2	2	(0.3)
Rana utricularia	RANUTR	1	0	0	1	(0.1)
Salamanders						
Eurycea multiplicata	EURMUL	0	2	3	5	(0.7)
Ambystoma opacum	AMBOPA	0	1	0	1	(0.1)
Ambystoma talpoideum	AMBTAL	0	1	0	1	(0.1)
Turtles						
Terrapene carolina	TERCAR	1	0	1	2	(0.3)
Terrapene ornata	TERORN	0	1	0	1	(0.1)
Lizards						
Sceloporus undulatus	SCEUND	80	54	45	179	(28.3)
Scincella lateralis	SCILAT	16	28	34	78	(12.3)
Cnemidophorus sexlineatus	CNESEX	48	5	2	55	(8.7)
Eumeces fasciatus	EUMFAS	8	20	6	34	(5.4)
Eumeces anthracinus	EUMANT	9	8	6	23	(3.6)
Eumeces laticeps	EUMLAT	10	3	4	17	(2.7)
Anolis carolinensis	ANOCAR	5	4	0	9	(1.4)
Snakes						
Agkistrodon contortrix	AGKCON	4	4	4	12	(1.9)
Coluber constrictor	COLCON	6	3	2	11	(1.7)
Thamnophis proximus	THAPRO	2	1	2	5	(0.8)
Storeria dekayi	STODEK	4	0	1	5	(0.7)
Elaphe guttata	ELAGUT	3	1	0	4	(0.6)
Thamnophis sirtalis	THASIS	0	1	2	3	(0.4)
Storeria occipitomaculata	STOOCC	1	0	2	3	(0.4)
Carphophis amoenus	CARVER	2	1	0	3	(0.4)
Masticophis flagellum	MASFLA	1	0	2	3	(0.4)
Lampropeltis triangulum	LAMTRI	2	0	1	3	(0.4)
Virginia valeriae	VIRVAL	1	0	1	2	(0.3)
Cemphora coccinea	CEMCOC	2	0	0	2	(0.2)
Heterodon platyrhinos	HETPLA	1	0	1	2	(0.3)
Diadophis punctatus	DIAPUN	1	0	1	2	(0.3)
Opheodrys aestivus	OPHAES	1	1	0	2	(0.3)
Tantilla gracilis	TANGRA	1	1	0	2	(0.3)
Elaphe obsoleta	ELAOBS	1	0	0	1	(0.1)
Sistrurus miliarius	SISMIL	1	0	0	1	(0.1)
Total					633	(100.0)

[a] Herpetological nomenclature follows Connant and Collins 1998.

abundance of woody debris. Clearcuts had scanty leaf litter, reduced canopy cover, and low pine and hardwood basal areas. Control and selectively harvested stands were closely grouped along CCA1 to the right and shared several habitat characteristics including greater litter depth, greater canopy cover, and greater pine and hardwood basal areas. In turn, selectively harvested and control stands differed along CCA3 (fig. 3b) due to greater herbaceous cover and more large, woody debris in selectively harvested stands; conversely, control stands had a higher proportion of woody plant cover. These habitat differences are apparent from the variable means (table 3).

The most distinct separation of species groups (fig. 4) was between reptiles and amphibians, with reptiles predominating in clearcuts and amphibians being most abundant in

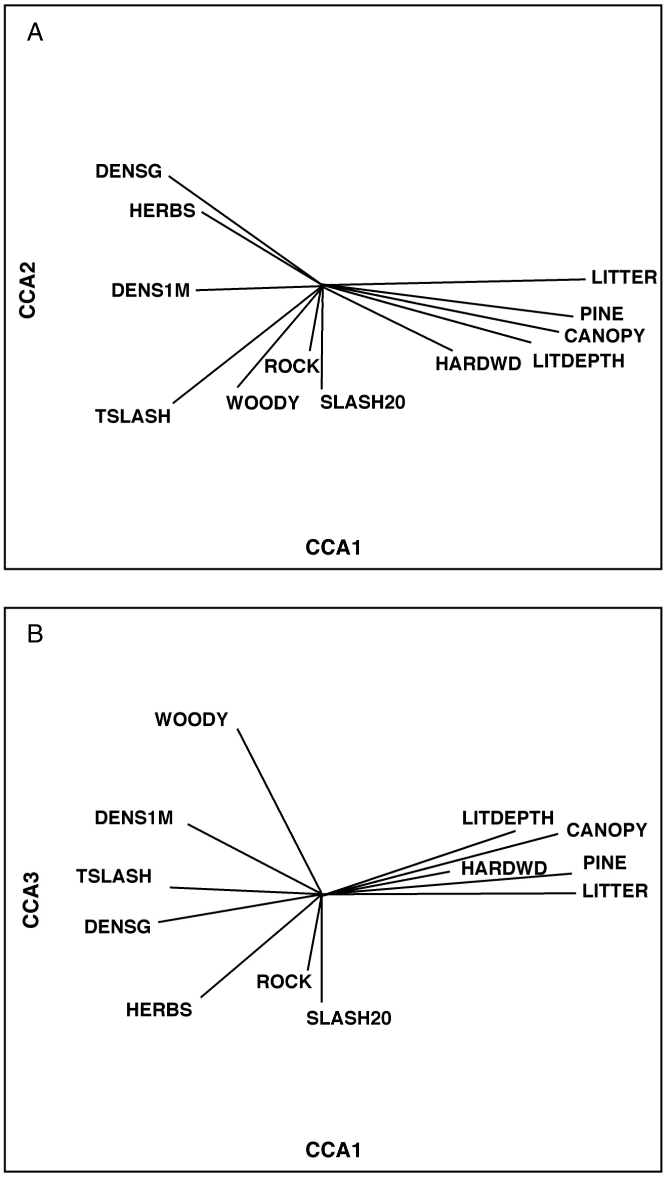

Figure 2—Canonical correspondence analysis (CCA) ordination of habitat variables: CANOPY = canopy cover, DENSG = horizontal cover at ground level, DENS1M = horizontal cover at 1 m above the ground, HARDWD = hardwood basal area, HERBS = herbaceous cover, LITDEPTH = litter depth, LITTER = leaf litter cover, PINE = pine basal area, ROCK = exposed rock, SLASH20 = woody debris with a diameter >20 cm, TSLASH = total woody debris, and WOODY = woody plant cover.

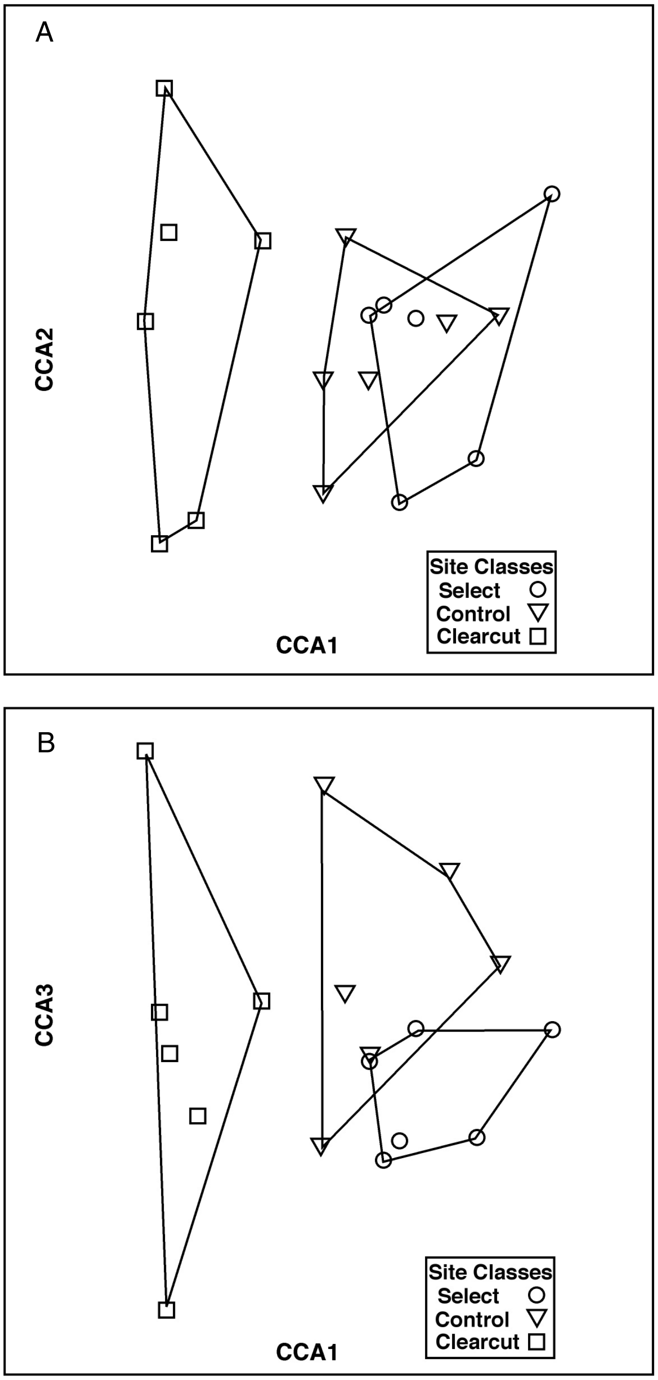

Figure 3—Canonical correspondence analysis (CCA) ordination of sample sites against (A) axes 1 and 2, and (B) axes 1 and 3.

forested stands. Frequencies of captures by species groups showed the same pattern (table 2).

Although not abundant in control or selectively harvested stands, salamanders were never observed in either clearcut. *Eurycea multiplicata*, the most abundant of the three salamander species (table 2), was not strongly associated with any habitat variable (fig. 5). All salamanders and most frogs and toads were collected in forested areas. *Gastrophryne carolinensis* and *Bufo americanus* were the most abundant frogs and toads (table 2) and were strongly associated with the higher canopy cover and litter depth of the two forested treatments (fig. 5). Both species were most common in selectively harvested stands (table 2).

The most commonly encountered snakes were *Agkistrodon contortrix* and *Coluber constrictor* (table 2). *Agkistrodon contortrix* was found equally in all treatments, whereas we caught more *C. constrictor* in clearcuts. These species occurred in a broad range of habitats within both forested

Table 3—Means and standard deviations (in parentheses) of habitat variables by treatment[a]

Habitat variable	Clearcut		Selective harvest		Control	
Slope (percent)	15.0	(1.8)	12.0	(4.5)	12.0	(2.5)
Pine basal area (m^2/ha)	0.0	(0.0)	16.1	(3.8)	19.0	(3.8)
Hardwood basal area (m^2/ha)	0.2	(0.4)	2.3	(1.6)	7.3	(3.6)
Canopy cover (percent)	7.0	(12.1)	83.0	(7.2)	97.0	(2.0)
Litter depth (cm)	1.0	(0.9)	3.0	(0.5)	3.0	(1.5)
Litter coverage (percent)	25.0	(8.9)	97.0	(2.3)	99.0	(3.2)
Herbaceous cover (percent)	42.0	(14.2)	32.0	(12.6)	7.0	(6.6)
Woody plant cover (percent)	27.0	(24.8)	16.0	(6.2)	17.0	(8.0)
Total woody debris (percent)	20.0	(9.5)	10.0	(3.9)	4.0	(2.1)
Large woody debris (percent)	6.0	(5.8)	4.0	(4.6)	3.0	(2.5)
Exposed rock cover (percent)	3.0	(3.3)	3.0	(2.6)	2.0	(3.4)
Horizontal cover at ground level (percent)	87.0	(8.0)	69.0	(26.1)	31.0	(12.8)
Horizontal cover at 1 m (percent)	63.0	(27.4)	42.0	(22.6)	19.0	(14.6)

[a] Each value is based on a sample size of 36 measurements.

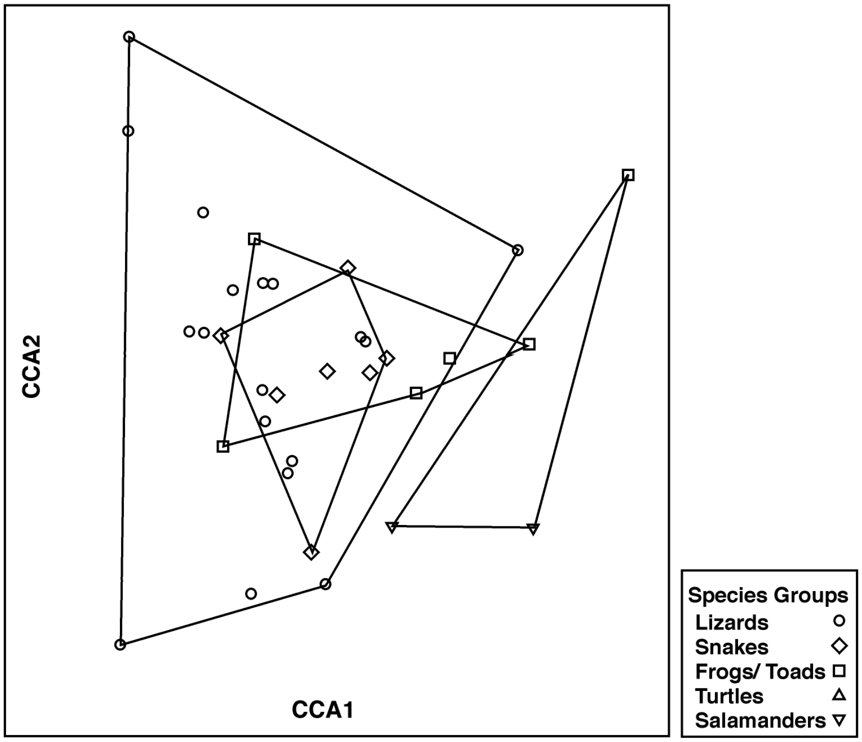

Figure 4— Canonical correspondence analysis (CCA) ordination of species groups, which can be superimposed on figures 2A and 3A in order to interpret patterns of community composition along silvicultural treatments and environmental gradients.

and open, grassy areas. Both species were associated with large, woody debris, woody vegetation, and exposed rock (fig. 5).

Thamnophis spp. were encountered mostly within controls and were generally observed near water, while both *Elaphe guttata* and *Storeria dekayi* were commonly observed within the clearcut stands (table 2). *E. guttata* was strongly associated with dense, herbaceous ground cover (fig. 5).

Lizards were the most abundant taxon (table 2), occupying most habitats (fig. 5). The most abundant species, *Sceloporus undulatus* (n = 179), was found in a wide variety of habitats and is therefore found near the center of the ordination (fig. 5). It was most common in clearcuts, but was also found in good numbers in the other treatments (table 2). *Scincella lateralis* and *Eumeces fasciatus* were most commonly encountered in forested areas (table 2) in association with abundant litter (fig. 5). *Cnemidophorus sexlineatus*

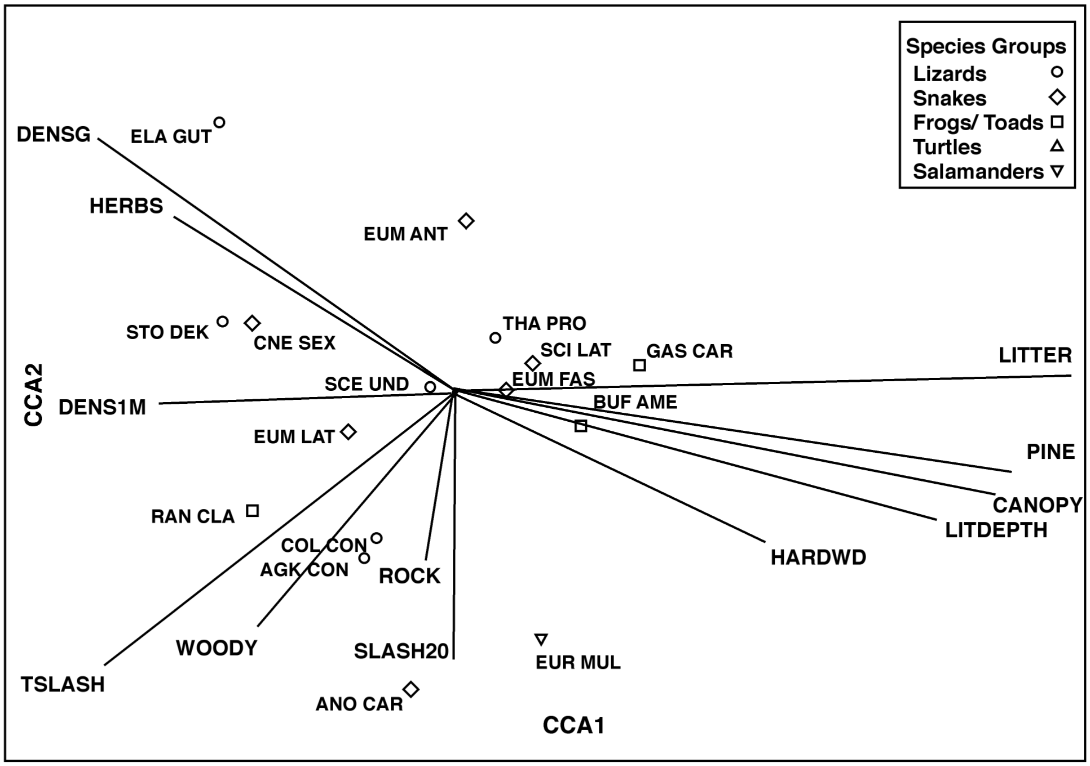

Figure 5— Canonical correspondence analysis (CCA) ordination of species and habitat variables. See table 2 for a key to species' acronyms.

and *Eumeces laticeps* were more prevalent in clearcuts (figs. 3 and 5), while *Eumeces anthracinus* was not clearly associated with any of the treatments (table 2). *Anolis carolinensis* was common in clearcut and selectively harvested stands, but was never found in control stands (table 2). *Anolis carolinensis* was associated positively and *Eumeces anthracinus* negatively to an abundance of large, woody debris.

DISCUSSION

Herpetofaunal communities differed significantly among treatments. Generally, microhabitat preferences of species explain these differences, with reptiles and amphibians responding predictably to gross changes in habitat structure among treatments. Because of increased insolation, higher ground temperatures, and higher evaporative water loss, fewer amphibians would be expected in young clearcuts (Bennett and others 1980; Geiger 1971; Petranka and others 1993, 1994). Although there was considerable overlap of taxa among habitats, our results suggested that amphibians generally favored forested areas over clearcuts. In particular, the two most common amphibians, *Gastrophryne carolinensis* and *Bufo americanus*, were strongly correlated with litter depths.

Unlike amphibians, reptiles preferred the open sunny habitats of the young clearcuts. Of the variables examined, horizontal cover and presence of woody debris (positively), and canopy cover and litter (negatively) seemed to be the most important factors determining reptile species composition within clearcuts. In loblolly-shortleaf pine (*Pinus taeda* and *P. echinata*) stands of east Texas, Whiting and others (1987) also found that vegetative cover and the degree of deciduous litter accumulation strongly influenced herpetofaunal communities. We found that *E. guttata* and *Cnemidophorus sexlineatus*, two grassland species, were both positively associated with dense, herbaceous ground cover and negatively with canopy cover. Surprisingly, *Eumeces laticeps*, an arboreal lizard, was strongly associated with woody plant cover and was more abundant in clearcuts than in either of the forested treatments. Enge and Marion (1986) found populations of *E. laticeps* to be reduced within clearcuts.

Some reptiles (especially lizards) may be attracted to recent clearcuts because the dense, low-growing vegetation provides an abundance of perching sites. For example, *Anolis carolinensis* was positively associated with woody plant cover and large, woody debris, habitat features largely absent from control stands. *Cnemidophorus sexlineatus*, a cursorial lizard, often inhabits early-succession habitats, shrubby hillsides, and open, grassy areas (Collins 1993, Webb 1970). Like us, Enge and Marion (1986) found this lizard favored the most intensively disturbed clearcuts.

A greater abundance and diversity of prey (invertebrates, birds, and small mammals) may contribute to higher abundances of reptiles (especially snakes) within clearcuts. A sharp increase in small mammal densities could attract large snakes such as *E. obsoleta* and *E. guttata*, which were found primarily in clearcuts. The Fulvous Harvest Mouse, *Reithrodontomys fulvescens*, Southern Short-tailed Shrew, *Blarina carolinensis*, Golden Mouse, *Ochrotomys nuttalli*, and mice, *Peromyscus* spp. were commonly captured in our pitfall traps in the clearcuts, while only *Peromyscus* spp. were captured in the other two treatments.

Perhaps the most significant limitation of this study is pseudoreplication (Hurlbert 1984). The three arrays within each treatment replicate were not spatially independent sites. Given adequate resources, it would be best to have multiple, spatially independent replications of each treatment. We attempted to limit this problem by separating sample sites by 100 m, but readers should use caution in interpreting the results.

Some species (e.g., *Thamnophis proximus*, *Storeria dekayi*, and *Eurycea multiplicata*) were not associated with any of the habitat variables we measured. Potentially important variables for future studies include invertebrate and small-mammal prey densities, microclimate, and proximity of sample sites to water. The latter two should aid in predicting occurrence of most amphibians (especially semiaquatic salamanders such as *E. multiplicata* and *Desmognathus brimleyorum*) (Crosswhite and others 1998), as well as reptiles (e.g., *T. proximus* and *T. sirtalis*) known to inhabit riparian areas or sloughs (Collins 1993, Webb 1970).

This and other studies (Crosswhite and others, in press; Dodd 1991; Gibbons and Bennett 1974; Gibbons and Semlitsch 1982; Jones 1986) suggest that some species (e.g. turtles and tree frogs) are not effectively sampled using pitfall and funnel traps. These species might be more effectively sampled using alternate techniques such as quadrat sampling, aural surveys for frogs and toads, or artificial habitat, (i.e., cover boards, frog houses, and artificial pools) (Heyer and others 1994).

Reptiles are generally favored by more open canopies and denser understory, as well as the presence of woody debris or rocky outcroppings. Most habitat characteristics determining herpetofaunal community composition are ultimately dependent upon stand age and the degree of site disturbance.

Reptiles and amphibians play significant roles in many ecosystems. They can be important components of the food web and may contribute substantially to community biomass (Burton and Likens 1975, Pough and others 1987). Furthermore, because amphibians are often habitat specialists with restricted distributions, they may be valuable indicators of ecosystem health and stability. We hope our findings will aid land managers in protecting reptile and amphibian habitat.

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Poster Session

A 5-YEAR ASSESSMENT OF SHORTLEAF PINE AND HARDWOOD SPROUTS RELATIVE TO THREE METHODS OF HARDWOOD CONTROL IN THE ARKANSAS OZARKS

Michael D. Cain[1]

Abstract—Compared with untreated checks, manual hardwood control and herbicide injection of hardwoods facilitated the development of direct seeded shortleaf pine (*Pinus echinata* Mill.) regeneration following a single-tree selection harvest in a mature natural stand of shortleaf pines in northwest Arkansas. Five years after hardwood control, shortleaf pine seedlings on treated plots were 215 percent taller ($P = 0.02$) and 242 percent larger ($P = 0.01$) in groundline diameter than pine seedlings on check plots. Resprouting hardwoods on herbicide injection plots were 42 percent shorter ($P < 0.01$) and had 72 percent less ($P = 0.01$) crown area compared with those on manual control plots.

INTRODUCTION

For regulated uneven-aged silviculture of shortleaf pine (*Pinus echinata* Mill.), the majority of stems in a stand should be in the seedling, sapling, and pulpwood size classes to allow for a continuous progression of trees into more valuable saw logs. To accomplish that objective, the hardwood component must be periodically controlled; otherwise, shade-intolerant pine seedlings that develop in the understory will most likely die (Cain 1987, 1988b).

Alternative methods for vegetation management include prescribed fire, mechanical methods, manual methods, herbicide treatments, and biological methods. In the Final Environmental Impact Statement for the Ozark/Ouachita Mountains (U.S. Department of Agriculture 1990), the preferred alternative for vegetation management specified an increase in the use of manual methods and a decrease in the use of herbicides and soil-disturbing mechanical methods. When herbicides are used, priority is to be given to chemicals and application methods that pose minimum risks to humans, wildlife, and nontarget plants.

This study was initiated to demonstrate uneven-aged silviculture of shortleaf pine by converting a mature even-aged stand using single-tree selection. Objectives were (1) to compare the cost and effectiveness of hardwood control by manual methods with that of stem-injected herbicides and (2) to monitor the establishment and development of shortleaf pine regeneration and hardwood regrowth following manual control and herbicide control of the hardwood component.

MATERIALS AND METHODS

Study Area

The study was located in Johnson County, AR, on the Bayou Ranger District of the Ozark National Forest. Plots were situated on a south-facing slope, and the elevation ranged from 760 to 920 feet. Soils are Nella (Typic Paleudult) and Mountainburg (Lithic Hapludult), gravelly or stony, fine sandy loam (U.S. Department of Agriculture 1977). Annual precipitation averages 46 inches with seasonal extremes being wet springs and dry autumns and winters.

The south-facing slope contained a mature stand of shortleaf pines with basal area averaging 85 square feet per acre in trees > 4.5 inches in diameter at breast height (d.b.h.). Most pines were from 14 to 18 inches d.b.h., and the understory was devoid of pine regeneration. Overstory and midstory hardwood basal area (> 4.5 inches d.b.h.) averaged 32 square feet per acre. There had been no apparent forest management activity in the stand during the last 20 years. Site indices at 50 years for shortleaf pine were determined by slope position in accordance with U.S. Department of Agriculture (1976): upper slope = 59 feet, midslope = 64 feet, and lower slope = 69 feet.

Study Installation

Twelve plots of 0.5 acre (147.6 by 147.6 feet) each, with interior subplots of 0.25 acre (104.4 by 104.4 feet), were established in the summer of 1990. The experimental design was a randomized complete block with three replications of four treatments. Blocking was based on topographic position—upper (22-percent slope), middle (26-percent slope), and lower (10-percent slope). Within each interior subplot, 25 systematically spaced sample points were permanently established to serve as centers of circular 1-milacre and 2-milacre quadrats for monitoring seedbed disturbance, pine seedling and sapling development, and nonpine competition.

All merchantable-sized pines (> 4.5 inches d.b.h.) were inventoried by 1-inch d.b.h. classes on a plot-by-plot basis. These data were used to determine the allowable cut based on the basal area–maximum diameter–quotient (BD*q*) technique (Farrar 1984). The BD*q* technique was applied according to three guidelines, in order of importance: (1) a basal area of 60 square feet per acre was to be left in merchantable-sized pines after harvest; (2) all pines larger than a maximum d.b.h. of 18 inches were to be cut, but some larger pines were retained to achieve the desired basal area; and (3) the residual diameter distribution should approach a balanced uneven-aged structure, characterized by a constant ratio (q) between the number of trees in succeeding diameter classes ($q = 1.22$ for 1-inch d.b.h. classes).

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Pine harvesting on study plots was completed in late June 1990. Cut pines were removed tree-length using an articulated rubber-tired skidder. Merchantable pines on an additional 40 acres surrounding the research plots were also marked for harvest using single-tree selection guidelines, but harvesting was delayed until the summer of 1991.

Treatments

Hardwood control was accomplished in early September 1990 by an independent contractor. An untreated check was compared with three methods of postharvest hardwood control as follows:

Check—No hardwood control, but there was disturbance from logging.

Herbicide injection—All hardwood trees and shrubs that were ≥ 1.0 inch groundline diameter (g.l.d.) were stem injected near the root collar with Garlon® 3A (triclopyr) at the rate of 0.02 ounce of undiluted herbicide per incision and one incision per inch of g.l.d. This treatment was accomplished with Jim-Gem® tree injectors.

Manual hardwood control—All hardwood trees and shrubs that were ≥ 1.0 inch g.l.d. were manually cut using chain saws. Stump heights averaged < 1.0 foot.

Manual hardwood control plus release—Hardwoods were controlled using the same method as the previous treatment, but pine regeneration was manually released during the fourth growing season (June 1994) after the initial treatment. For release, brush axes and lopping shears were used to cut competing vegetation near the groundline but only within 4 feet of established pine seedlings. If no pine seedlings were visually observed as workers traversed each plot, then no vegetation was cut in the release treatment.

In order to assess the cost of herbicide and manual treatments for controlling hardwoods, the time required to inject or manually fell the hardwoods was recorded, and the volume of injected herbicide was measured. Six 0.5-acre plots were assessed for herbicide injection cost, and six plots were assessed for chain saw felling cost. To assess labor cost, the minimum wage in 1990 was doubled to $8.50 per hour to account for vendor contract fees. The cost of Garlon herbicide was based on the retail price in 1991 ($169 for 2.5 gallons). The cost of chain saw operation was based on dollar amounts taken from Miller (1984) with an addition of 5 percent annually for 6 years to account for inflation and was determined to be $2.76 per hour of chain saw use.

On the 40-acre nonresearch area that surrounded study plots, residual hardwoods > 1 inch d.b.h. were controlled after pine harvesting was completed by using an edge-to-edge cut-surface treatment at waist height and applying Accord® (glyphosate) herbicide at recommended rates (25 to 50 percent concentration).

Pine Seed Crops

Natural pine seed fall was monitored during 3 consecutive seed years (1990-91, 1991-92, and 1992-93), from October through January, using 4 seed traps on each of the 12 plots. The four interior subplot corners were used as monumentation points for the seed traps. Individual seed traps had an opening of 0.9 square foot (Cain and Shelton 1993). All 3 seed years were judged to be complete failures, because the number of potentially viable seeds ranged from 1,000 to 3,000 per acre per year. According to Liming (1945), 40,000 viable shortleaf pine seeds per acre are purported to be necessary during any one seed year for adequate regeneration under natural conditions.

To ensure regeneration of the area, Bayou Ranger District personnel obtained from an Ozark Mountain source shortleaf pine seeds that had been in cold storage. Using cyclone hand-seeders, district personnel broadcast the seeds onto the 12 study plots at the rate of 3 pounds per plot (6 pounds per acre) in mid-February 1991. These seeds were reported to average 28,100 seeds per pound and had a germination rate of 86 percent. A heavy sowing rate was used because the seeds were not treated with repellents to control predators or stratified before direct seeding. The recommended rate for broadcast sowing of repellent-treated, stratified, shortleaf pine seeds is 0.48 pound per acre (Ezell 1988).

One year after direct seeding, the 40-acre area surrounding the research plots was hand-planted with 1-0, genetically improved, bare-root shortleaf pine seedlings from an eastern Oklahoma seed source. These seedlings were outplanted with hoedads on a 12- by 12-foot spacing (302 trees per acre). To monitor the development of planted seedlings, an additional 0.2-acre plot was established contiguous to each of the herbicide injection plots in the main study. During autumn of 1992, 25 planted shortleaf pine seedlings were randomly selected for measurement within each planted plot and tagged for identification.

Measurements and Data Analysis

Following harvest and hardwood control, seedbed conditions were ocularly estimated to the nearest 10 percent on each of the twenty-five 1-milacre sample quadrats per plot. Eight disturbance codes were used in that assessment (Cain 1993), but only the percentage of forest floor covered by manually cut hardwood slash is presented here.

In the winter of 1995-96, 5 years after hardwood control, merchantable-sized pines (> 4.5 inches d.b.h.) on each 0.5-acre plot were remeasured to 0.1 inch at d.b.h. for calculation of basal area. Pine seedlings were counted within each of twenty-five 1-milacre sample quadrats per interior subplot. The two tallest (dominant) pine seedlings (stems ≤ 0.5 inch d.b.h.) within each milacre quadrat were measured for total height to 0.1 foot and g.l.d. to 0.04 inch. Within a 2-milacre circular plot on even-numbered quadrats, pine saplings (stems > 0.5 inch but < 4.6 inches d.b.h.) were measured, but only two saplings were present across all 12 plots 5 years after harvest. Within the twenty-five 1-milacre quadrats per plot, ground coverage was ocularly estimated to the nearest 10 percent for herbaceous vegetation (forbs, grasses, semiwoody plants, and vines) and for merchantable and submerchantable pines and hardwoods (including shrubs) by vertical projection of foliar cover to the ground.

During the winter measurements, hardwood rootstocks of seedling size were counted within each of the twenty-five 1-milacre quadrats per plot. A rootstock was comprised of either single or multiple stems (clump) of seedling size, which obviously arose from the same root system. The tallest seedling-sized hardwood was identified by species and measured for total height to 0.1 foot and for crown width at the widest axis and perpendicular to that axis to 0.1 foot. Within the 2-milacre quadrats at even-numbered sample points, hardwood saplings were counted by 1-inch d.b.h. classes, and the dominant hardwood sapling was identified by species.

For residual hardwoods that were > 4.5 inches d.b.h., a 100-percent inventory was conducted within each 0.5-acre plot. The numbers of hardwood stems were recorded by 1-inch d.b.h. classes and were categorized into four species groups: red oaks (principally *Quercus falcata* Michx., *Q. rubra* L., and *Q. velutina* Lam.), white oaks (*Q. alba* L. and *Q. stellata* Wangenh.), hickories (*Carya* spp.), and other hardwoods.

On the three plots established for monitoring planted pine development, surviving crop seedlings were also measured in the winter of 1995-96 to obtain total height to 0.1 foot and g.l.d. to 0.04 inch. On these plots, merchantable-sized pines were measured to 0.1 inch d.b.h. for calculation of basal area.

Analysis of variance was used to evaluate treatment effects on measured variables. Percentage values for quadrat stocking and ground coverage were compared following arcsine transformation. Treatment means were compared using orthogonal contrasts. Analysis of variance was also used to compare the mean size of planted pines with that of dominant seeded pines on herbicide injection plots in the main study.

RESULTS AND DISCUSSION

Cost of Postharvest Hardwood Control

Averaged over an area of 3 acres for each method of hardwood control, the initial cost of manually felling the hardwoods was half the cost of tree injection (Cain 1993). Chain saw felling was accomplished at a cost of $0.07 per treated stem as compared with $0.14 per stem for herbicide injection. For the injection treatment, Garlon accounted for 78 percent of the cost, with labor at 22 percent. For manual hardwood control, labor was the highest cost factor at 79 percent, and chain saw use accounted for 21 percent of the cost. Expenses incurred during the fourth growing season for release of established pine seedlings averaged $68 per acre, thereby increasing total cost for manual hardwood control in that treatment to $100 per acre and exceeding the cost of herbicide injection by $37 per acre.

During initial hardwood control, all hardwoods > 1.0 inch g.l.d. were treated, even if they were of merchantable size. On an operational scale, merchantable hardwoods should be sold and removed from the site, which would reduce the cost of postharvest hardwood control.

Postharvest Seedbed Condition

Cut-and-leave hardwood control techniques result in too much detritus being left on the site, and slash cover has been shown to impede natural pine regeneration (Trousdell 1950). In the present study, a preliminary analysis of post-treatment seedbed conditions indicated that, on the six plots subjected to manual hardwood control, 88 percent of sampled 1-milacre quadrats contained hardwood slash, and ground coverage from felled hardwoods averaged 62 percent of the area within those quadrats. Because such heavy slash restricted accessibility and was considered to be unacceptable for pine seedling establishment, the slash was removed from within plot boundaries if > 4.0 inches in diameter.

Seeded Pine Regeneration

In uneven-aged silviculture, 200 submerchantable-sized pines per acre are considered to be the minimum for adequate density, with 50-percent milacre stocking being optimum (Cain 1991). The preharvest inventory revealed no pine seedlings, but five growing seasons after direct seeding, pine seedling density averaged 1,164 stems per acre across all treatments. Herbicide injection plots had the most ($P = 0.03$) pine seedlings compared with other treatments; even so, all plots averaged > 500 seedlings per acre (table 1). Hardwood control plots also had better milacre stocking ($P = 0.01$) of pine seedlings than check plots, and milacre stocking on herbicide injection plots was better ($P = 0.01$) than on manual control plots. Herbicide injection of hardwoods was also the only treatment to result in > 50-percent milacre stocking of pine seedlings.

With direct seeding, the study area was successfully regenerated. However, stand structure was a long way from a well-regulated condition. Results from uneven-aged management of shortleaf pines in the Ouachita Mountains of Arkansas suggest that old, even-aged sawtimber stands are the most difficult to convert to uneven-aged structure and will require the longest conversion period of any existing stand type (Murphy and others 1991).

Five growing seasons after establishment, dominant pine seedlings averaged 1.90 feet tall and 0.34 inch g.l.d. (table 1). Dominant seedlings on hardwood control plots were 215 percent taller ($P = 0.02$) and 242 percent larger in g.l.d. ($P = 0.01$) than those on check plots (table 1). There was no difference ($P > 0.05$) in total height or g.l.d. for dominant pine seedlings on herbicide injection plots versus manual control plots.

Planted Pine Regeneration

In January 1992, 1 year after direct seeding of the plots in the main study, the nonresearch areas surrounding the main study plots were hand-planted with 1-0 bare-root shortleaf pine seedlings. At the time of remeasurement in the winter of 1995-96, both the dominant, direct-seeded shortleaf and the planted shortleaf were assumed to be 5 years old from seed. At that time, merchantable pine basal area on planted plots averaged 69 square feet per acre and did not differ ($P = 0.91$) from the 70 square feet per acre on herbicide injection plots in the main study. Therefore, the negative effect of overstory pines on the development of pine regeneration should have been the same regardless of establishment technique.

Table 1—Status of shortleaf pine regeneration 5 years after hardwood control

Hardwood control and orthogonal contrasts	Density	Milacre stocking^a	Total height^b	G.l.d.^{b}
	stems/ac	*percent*	*feet*	*inches*
1. Check	586	20	0.73	0.12
2. Herbicide injection	1,987	59	2.50	0.47
3. Manual hardwood control	961	41	2.04	0.35
4. Manual hardwood control, plus release	1,121	32	2.35	0.42
Mean square error	236,578	0.011	0.59	0.02
PR > F^c	0.06	0.01	0.10	0.06
	Probabilities of a greater F-ratio			
1 vs 2 + 3 + 4	0.06	0.01	0.02	0.01
2 vs 3 + 4	0.03	0.01	0.59	0.37
3 vs 4	0.70	0.28	0.64	0.51

G.l.d. = groundline diameter.

[a] Based on the presence of at least one pine seedling per 1-milacre quadrat.

[b] Mean total height and groundline diameter of the two dominant pine seedlings per 1-milacre quadrat.

[c] The probability of obtaining a larger F-ratio under the null hypothesis.

Planted shortleaf seedlings averaged 4.44 feet in height and were 78 percent taller ($P < 0.01$) than the dominant seeded shortleaf seedlings on herbicide injection plots in the main study. At 0.92 inch g.l.d., planted shortleaf seedlings were also 96 percent larger ($P = 0.01$) than dominant seedlings on herbicide injection plots. Survival of these planted shortleaf pines averaged 88 percent.

Because of a natural seed crop failure during the winter after harvest, direct seeding was chosen in the main study to simulate natural seed fall. Although no cost comparison was made between planting and direct seeding, 5-year results suggest that planting of shortleaf seedlings was superior to direct seeding in terms of growth response after controlling hardwoods by herbicide injection.

Competing Vegetation

Five years after hardwood control, density of seedling-sized hardwoods averaged nearly 3,700 rootstocks per acre (table 2). Although manual control plots averaged 32 percent more hardwood rootstocks of seedling size than plots that were treated by herbicide injection, treatment differences were not statistically significant ($P = 0.20$). However, dominant hardwood sprouts of seedling size were 73 percent taller ($P < 0.01$) on manual control plots than on herbicide injection plots. In addition to being taller, these dominant sprouts had crowns that averaged 253 percent larger ($P = 0.01$) than those on injected plots (table 2). Obviously, herbicide injection was more effective than chain saw felling by reducing the regrowth of hardwoods during the first 5 years after treatment.

Six woody species, or groups of species, predominated the seedling size class, and they accounted for 78-percent milacre stocking of all nonpine woody vegetation. These were *Cornus florida* L. (24 percent), *Quercus* spp. (20 percent), *Carya* spp. (13 percent), *Nyssa sylvatica* Marsh. (11 percent), *Vaccinium* spp. (6 percent), and *Acer rubrum* L. (4 percent). Milacre stocking was < 4 percent for other seedling-sized woody species.

Density of hardwood saplings averaged 556 stems per acre (table 2), and manual control plots had 633 percent more ($P = 0.02$) saplings than herbicide injection plots. Quadrat stocking of these sapling hardwoods averaged 42 percent across all plots (table 2). Manual control plots had 159 percent more ($P = 0.03$) quadrats stocked with hardwood saplings than did herbicide injection plots. This result again reflects the efficacy of herbicide injection versus chain saw felling when used to control hardwoods in pine management. As would be expected, the mean d.b.h. of hardwood saplings on check plots averaged 36 percent larger ($P = 0.03$) compared with those on treated plots (table 2). On check plots and manual control plots, *Cornus florida* L. were the predominant hardwoods of sapling size, whereas *Ulmus alata* Michx. saplings predominated on herbicide injection plots.

Five years after hardwood control, ground coverage by pine regeneration averaged only 5 percent (table 3). Herbicide injection resulted in greater ($P = 0.04$) ground coverage by pine seedlings compared with manual control treatments, but all hardwood control treatments had more ($P = 0.03$) pine seedling ground coverage than check plots. Ocular estimates of percentage ground cover from hardwoods of submerchantable size substantiates the differences in efficacy of the hardwood control treatments that were tested in this investigation. Ground coverage from these hardwoods averaged about 27 percentage points higher ($P < 0.01$) on manual control plots than on herbicide injection plots (table

Table 2—Status of nonpine competition 5 years after hardwood control

Hardwood control and orthogonal contrasts	Seedling-sized hardwoods			Sapling-sized hardwoods		
	Rootstock density	Total height[a]	Crown area[a]	Density	Quadrat stocking[b]	Weighted d.b.h.[c]
	no./ac	*feet*	*ft^2/rtstk*	*stems/ac*	*percent*	*inches*
1. Check	3,080	4.89	11.14	264	30	1.45
2. Herbicide injection	3,213	4.16	4.10	125	22	1.11
3. Manual hardwood control	4,693	7.01	14.68	986	67	1.03
4. Manual hardwood control, plus release	3,800	7.41	14.28	847	47	1.08
Mean square error	10.46×10^5	0.90	18.42	13.53×10^4	0.04	0.04
PR > F[d]	0.30	0.01	0.07	0.07	0.07	0.15
			Probabilities of a greater F-ratio			
1 vs 2 + 3 + 4	0.27	0.08	0.97	0.16	0.21	0.03
2 vs 3 + 4	0.20	< 0.01	0.01	0.02	0.03	0.73
3 vs 4	0.33	0.62	0.91	0.66	0.21	0.78

D.b.h. = diameter at breast height.

[a] Data from the dominant seedling-sized hardwood rootstock (rtstk) per 1-milacre quadrat.

[b] Based on the presence of at least one hardwood sapling per 2-milacre quadrat.

[c] Means weighted by number of hardwood saplings in the 1-, 2-, 3-, and 4-inch d.b.h. classes.

[d] The probability of obtaining a larger F-ratio under the null hypothesis.

Table 3—Ground cover by vegetative component 5 years after hardwood control

Hardwood control and orthogonal contrasts	Ground cover			
	Pine seedlings	Hardwoods <4.6 in. d.b.h.	Hardwoods >4.5 in. d.b.h.	Herbaceous vegetation
	- - - - - - - -	- - - - - - - - *percent*	- - - - - - - -	- - - - - - - -
1. Check	1.8	36.5	48.8	29.9
2. Herbicide injection	8.6	24.3	3.9	60.3
3. Manual hardwood control	4.6	52.7	0.0	58.8
4. Manual hardwood control, plus release	4.6	50.5	0.0	58.2
Mean square error	0.00048	0.006	0.008	0.032
PR > F[a]	0.05	0.01	< 0.01	0.14
		Probabilities of a greater F-ratio		
1 vs 2 + 3 + 4	0.03	0.22	< 0.01	0.03
2 vs 3 + 4	0.04	< 0.01	0.57	0.76
3 vs 4	0.99	0.73	1.00	0.97

in. = inches; d.b.h. = diameter at breast height.

[a] The probability of obtaining a larger F-ratio under the null hypothesis.

3). Ground coverage from merchantable-sized hardwoods averaged 48 percentage points higher ($P < 0.01$) on check plots compared with plots where these trees were controlled 5 years earlier (table 3). On check plots, the multistoried hardwood cover, in combination with overstory pines, contributed to almost complete shading of the forest floor.

Residual basal area in merchantable-sized hardwoods on check plots averaged 36 square feet per acre. These hardwoods ranged from 5 to 18 inches d.b.h. and averaged 116 trees per acre. Of that density, 43 percent was in the white oak group, 28 percent in hickories, 18 percent in the red oak group, and 11 percent in other hardwoods.

Five years after treatment, ground coverage from herbaceous vegetation averaged 52 percent (table 3). Because of their intolerance to shade, herbaceous plants had less (*P* = 0.03) ground cover on check plots than on hardwood control plots. In the first few years after establishment of pine regeneration, competition from herbaceous vegetation can be more detrimental to survival and growth of pine seedlings than woody competition, especially on good sites (site index > 85 feet at 50 years) where herbaceous ground cover can approach 100 percent, with multistoried layers (Cain 1988a). Given the combined ground cover from sprouting hardwoods and herbaceous vegetation in the present study, and in the absence of site disturbance, it is unlikely that additional pine regeneration will become established from natural seed fall on this site, even in better-than-average seed years.

MANAGEMENT IMPLICATIONS AND CONCLUSIONS

In order for shade-intolerant shortleaf pines to become established at recommended levels of density and quadrat stocking and grow at an acceptable rate for sustaining uneven-aged silviculture, removal of overtopping hardwoods is required. If a landowner wishes to retain some midstory and overstory hardwoods while managing for pine, then clumping of hardwoods would be more desirable than random distribution (Shelton and Murphy 1993). In fact, Leopold and others (1985) found that most hardwood species in a midwestern hardwood forest exhibited an aggregated distribution under natural conditions.

In the present investigation, herbicide injection of hardwoods after the pine single-tree selection harvest was more beneficial in terms of enhancing the establishment and growth of direct-seeded shortleaf pine regeneration when compared with untreated checks or manual hardwood control treatments. Followup manual release of established pine regeneration in the fourth growing season resulted in no measurable benefit in terms of competition control or pine seedling growth when compared with initial hardwood control by chain saw felling. The release treatment also negated the initial cost advantage of chain saw felling. If herbicide use is not an option on National Forest System lands because of public concerns, then manual girdling of the hardwood component is likely to result in fewer sprouts of smaller size than chain saw felling (Liming 1945, 1946).

During the first 5 years of this study, residual overstory pines grew at the rate of only 0.14 inch d.b.h. per year. Without ingrowth from submerchantable to merchantable size classes, data suggest that cycle cuts may be needed only every 10 to 12 years for conversion of mature even-aged shortleaf pine stands to uneven-aged structure on south-facing slopes where site conditions are similar to those tested here. This assumes that residual basal area averages 60 square feet per acre in merchantable-size pines after the initial harvest.

Although the evaluation of planted shortleaf pine seedlings was conducted only as a side test to the main study, the genetically improved, nursery-grown pines provided better stocking and outperformed dominant shortleaf seedlings that were established by seeding. Consequently, when converting mature even-aged shortleaf pine stands to uneven-aged structure, planting appears to be a viable regeneration alternative to direct seeding or natural seeding, especially at locations where natural seed crops are consistently below average.

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EFFECTS OF ALL-TERRAIN VEHICLES ON STREAM DYNAMICS

Anne Chin, Deven M. Rohrer, Daniel A. Marion, and J. Alan Clingenpeel[1]

Abstract—This paper reports preliminary results from research conducted in the Ouachita National Forest to assess the effects of all-terrain vehicle (ATV) trails on stream characteristics. The study focuses on the Wolf Pen Gap Trail that has been in use since 1991. We examine whether that the trail system has caused increased sediment input to and deposition within stream pools. We compared selected pool characteristics in two watersheds with ATV trails (Gap and Board Camp Creeks) to those in two control watersheds (Brushy and Caney Creeks). Analysis indicates that watersheds with ATV trails have pools with higher percentages of sands and fines, lower depths, and lower volumes. More research is needed to confirm that ATV trails in fact cause these impacts, to determine what factors control impact occurrence, and to better quantify impact magnitude.

INTRODUCTION

Previous research shows that road and trail crossings generally have negative impacts on the hydrology and geomorphology of forest streams. Roads and trails are active sources of sediment, and they increase peak flows in some cases (Reid and Dunne 1984, Wemple and others 1996). Heavy usage can accelerate erosion, compact soils, and decrease infiltration, leading to changes in discharge magnitude and timing, channel structure, sediment routing through forest streams, and habitat degradation (e.g., Brown 1994, Eckert and others 1979, Webb 1983). These impacts may be more pronounced in the case of all-terrain vehicle (ATV) trails, where users often develop improperly located trails in addition to designated ones.

In the Ouachita National Forest (ONF), the Wolf Pen Gap All-Terrain Vehicle Trail was created upon pre-existing roads and opened in 1991 for public use. The system consists of 67 km of designated trails that loop through an array of areas, including scenic Gap Creek and Board Camp Creek (fig. 1). Because of the prevalence for off-road exploration, and the ease with which ATVs can traverse rugged terrain, users have also developed a network of unauthorized trails. These trails totaled 28.2 km by 1997. Because these trails are undesigned, they can be especially erosive and potentially exacerbate any negative impacts of the planned trail system on channel integrity.

The ONF began monitoring the effects of the ATV trail system on water quality and stream characteristics in 1990. Data collection continued through 1999, wherein we also completed the mapping of the ATV trails and other roads in the study area. In this paper, we report preliminary results comparing pool characteristics between watersheds with and without ATV usage during the initial period following ATV trail construction and usage (1990-1992).

Figure 1—Authorized all-terrain vehicle trail in the Gap Creek watershed.

STUDY AREA

The study uses four basins within the ONF in Arkansas. The ONF encompasses much of the Ouachita Mountains, part of the Southern Interior Highlands of the United States (Smith 1989). The area features east-west trending ridges and valleys, which were formed by the erosion of tightly folded sedimentary rocks. The soils tend to be thin Inceptisols and Ultisols with predominantly forest cover. The vegetation is largely composed of loblolly and shortleaf pine mixed with scattered hardwoods. The region has a humid subtropical

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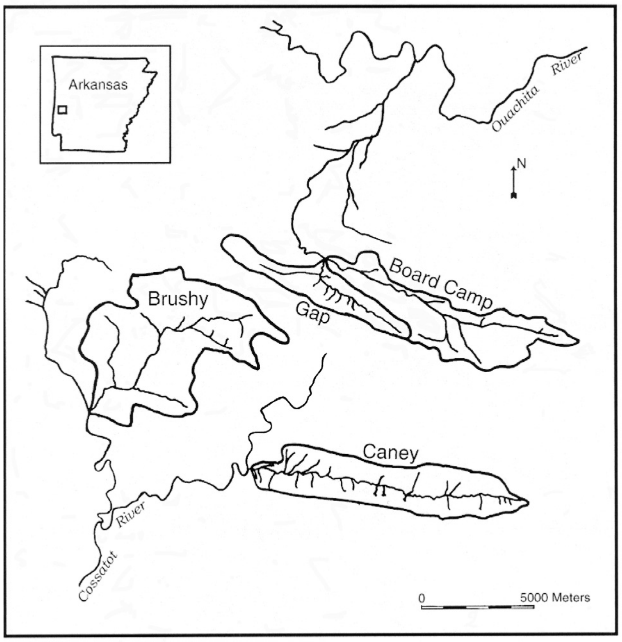

Figure 2—Study basins in the Ouachita National Forest, Arkansas.

climate characterized by warm summers and relatively cold and dry winters. Annual precipitation in Arkansas averages 124 cm, although orographic effects contribute to high rainfall variability in the study area.

The study basins of Board Camp, Gap, Brushy, and Caney Creeks are all located in the western portion of the ONF (fig. 2). The basins have similar geology, climate, soils, and vegetation and are representative of watersheds within the ONF. Board Camp and Gap Creeks flow generally westward and northward, draining eventually into the Ouachita River. Their watersheds contain the Wolf Pen Gap Trail, whereas ATV trails are absent in the Brushy Creek and Caney Creek basins. A road network does exist in the Brushy Creek watershed. However, unlike the situation with ATV trails, these roads only cross the streambed at culverts and designed fords. Caney Creek is the least disturbed of the study basins. Situated in the Caney Creek Wilderness Management Area, only hiking and horse trails are allowed. Both Caney and Brushy Creeks flow generally westward/southward and drain into the Cossatot River.

METHODS

The study approach focuses on comparing the stream characteristics of Board Camp Creek and Gap Creek (streams with ATV trails) with those of Brushy Creek and Caney Creek (reference streams without ATV trails). We measured physical characteristics of the study streams using Basin Area Stream Survey methods (BASS). BASS is a systematic procedure for inventorying the physical, chemical, and biological characteristics of forest streams (Clingenpeel and Cochran 1992). As part of the BASS, habitat types are identified according to McCain and others (1990); these include low- and high-gradient riffle, lateral scour pool, mid-channel pool, step pool, bedrock sheet, and run. Physical characteristics measured include bankfull width, water depth, substrate material size, and bank stability. These characteristics were used to compute parameters for evaluating ATV trail impacts: percent sands and fines (grains < 2.0 mm), pool depth, pool volume, and embeddedness. Complete habitat data are contained in Clingenpeel (1994). For this analysis, only those habitat types containing pools are examined and data from all pool habitat types are combined for each basin.

RESULTS AND DISCUSSION

Turbidity

Visual turbidity observations indicate that sediment inputs differ between the ATV-affected and control streams as a result of ATV trail usage. High turbidity levels were observed in surface runoff from ATV trails entering Gap and Board Camp Creeks during and after light rainstorms in the summer of 1999 (fig. 3). Pools below ATV trail crossings were no longer clear but muddy and sediment laden (fig. 4). Such

Figure 3—Sediment transport along all-terrain vehicle trails after rainfall, Gap Creek.

Figure 4—Example of turbid pool in Gap Creek at all-terrain vehicle crossing after rainfall.

observations indicate the erosive nature of the ATV trails, and they suggest that ATV trails may be significant sediment sources for streams within Gap and Board Camp Creek basins.

Percent Sands and Fines

Visual observations also indicate that the creeks affected by ATV trails are experiencing increased sedimentation. Figure 5 shows typical examples of pools within ATV-affected and unaffected basins. The larger amount of sands and fines in the affected basin is readily apparent. The larger fine-sediment fractions indicate that sediment inputs are large enough that the streams cannot readily transport these additional loads.

Measured percents confirm that sands and fines are substantially higher in stream pools within the impacted basins. Box plots in figure 6 show the higher values of percent sands and fines in Board Camp and Gap Creeks. The median value approaches 20 percent in both of these streams, about twice that of Brushy and Caney Creeks. The 25- and 75-percentile values range between 10 and 30 percent sands and fines for Board Camp and Gap Creeks, whereas in the reference creeks, they range from 0 to 10 percent. The 90-percentile values for Board Camp and Gap Creeks reach 50 percent and greater sands and fines. In contrast, for the streams without ATV trails, 90 percent of the pools have only 20 percent sands and fines or less.

Embeddedness

Embeddedness is a measure of the extent to which cobble-sized material is surrounded by finer materials. It is expressed as the percent of a cobble surrounded by (or embedded in) smaller grains; it is an additional descriptor of sediment load in streams.

The quantitative data for embeddedness do not show clear differences between ATV-affected and unaffected streams (fig. 7). Median values of embeddedness are about 30 to 35 percent for all four streams. The box plots show larger ranges

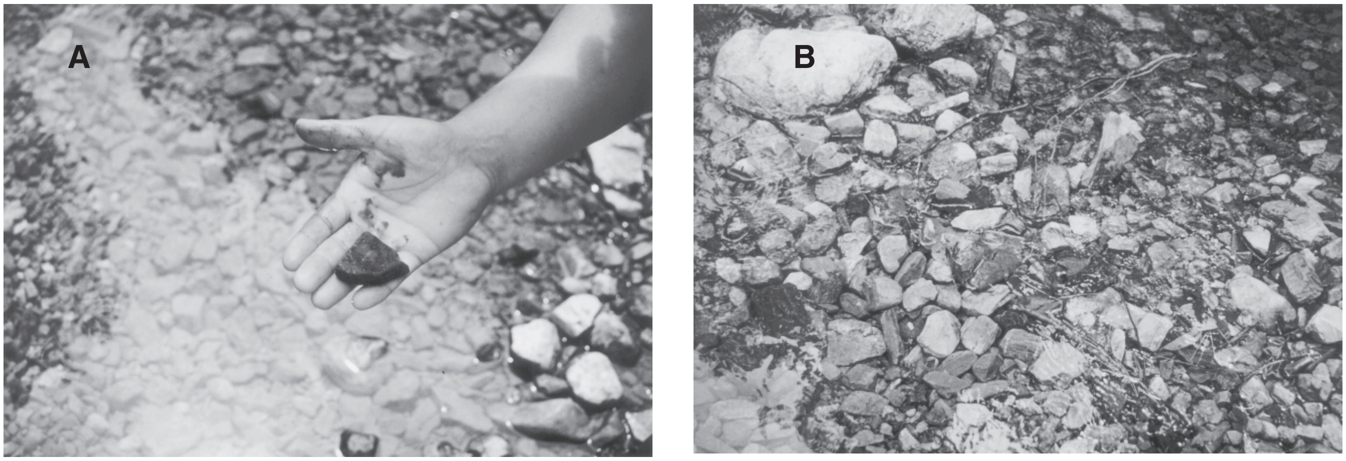

Figure 5—(A) fine sediment coats cobbles within pool in all-terrain vehicle affected basin, and (B) cobbles in Caney Creek pool are clearly visible through water.

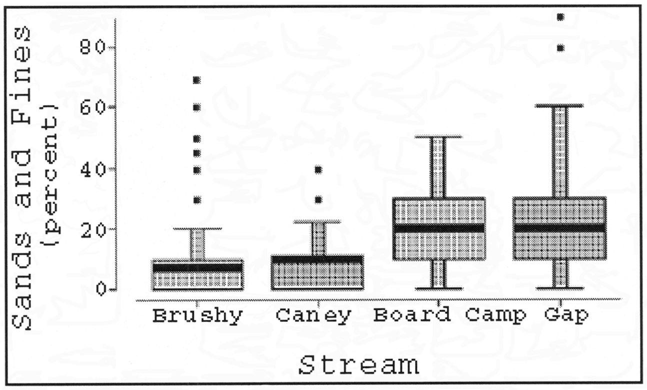

Figure 6—Box-plots illustrating percent sands and fines in pools of the four study streams. Median is represented by the thick centerline. The box encloses values between the 25- and 75-percentiles. Whiskers extend from the box to the 10- and 90-percentiles of the data. Points outside the whiskers are extreme values.

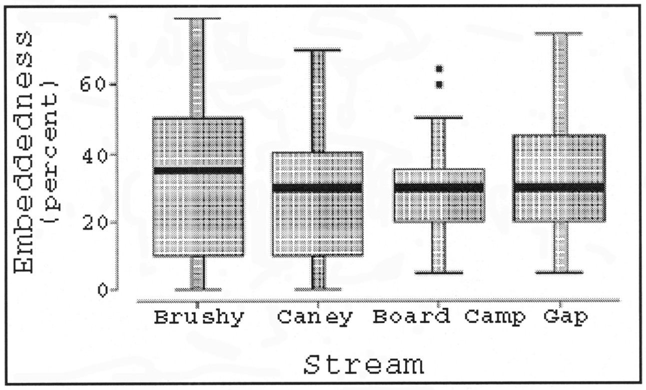

Figure 7—Box-plots of embeddedness for stream pools within the four study basins.

of embeddedness values for the creeks without ATV trails. Although Board Camp and Gap Creeks exhibit narrower ranges of embeddedness values, a somewhat larger proportion of the samples in these impacted streams have higher embeddedness values. For example, there are no samples in these creeks with < 10 percent embeddedness and 75 percent of their samples have values > 20 percent.

Embeddedness is a measure that has important implications for fish habitat. When embeddedness increases, the interstitial spaces around cobbles decrease, thereby decreasing the quantity and quality of habitat for aquatic macroinvertebrates and small fish such as darters. Therefore, any change in channel embeddedness, particularly if the change is an increase, can cause a series of complex chain reactions that may be difficult to clarify and manage effectively.

Pool Depth

Streams impacted by ATV trails are somewhat shallower than those in the reference watersheds (fig. 8). Median depths are 20 to 25 cm in Board Camp and Gap Creeks, about half the values of Brushy and Caney Creeks. The 75-percentile values for Board Camp and Gap Creeks are also lower than those of the reference streams.

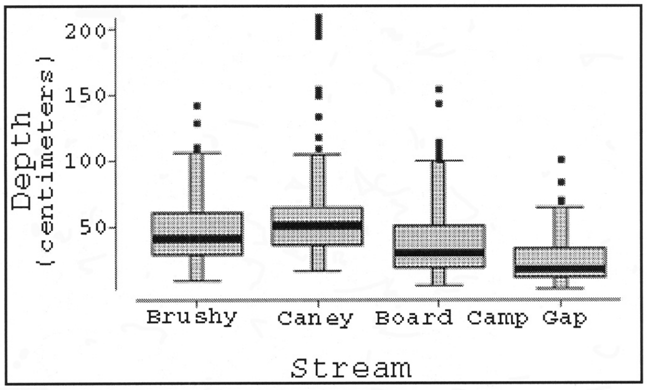

Figure 8—Box-plots of pool depths within the four study basins.

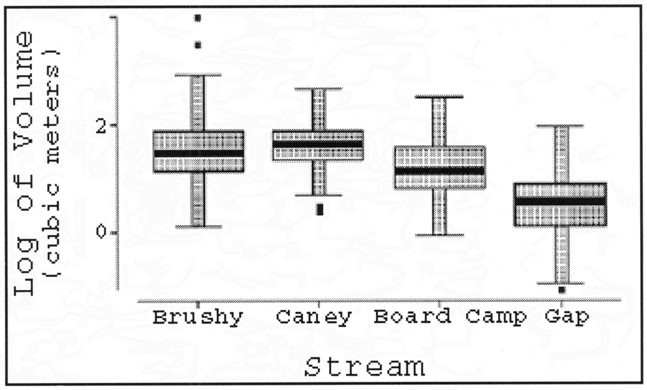

Figure 9—Box-plots of pool volumes within the four study basins.

Decreased depth in ATV-impacted streams is consistent with increased sedimentation. Depth is an indication of how viable pool habitat is during low flow or drought conditions. Shallow pools do not provide the cover of deep pools and are not as viable during drought. Therefore, these data suggest a possible decline in the overall health of stream ecosystems affected by ATV trails.

Pool Volume

Most water volume in Ouachita headwater streams is contained in pools. Because of the intermittent nature of these streams, pools serve as primary habitats for larger fish and as a refuge for all fish during low flow or drought conditions. Figure 9 shows pool volume is notably less for Board Camp and Gap Creeks. Decreased pool volume is an expected result if sediment inputs and deposition had increased in these streams. As with lowered pool depths, smaller pool volumes suggest possible impairment in the ecological functioning of the two creeks affected by ATV trails.

CONCLUSIONS

In summary, preliminary analysis of paired watersheds indicates that the Wolf Pen Gap Trail system appears to have had detectable negative impacts on the structure and habitat quality of stream pools. These impacts are reflected in an increase in fine sediments in pools along with a decrease in pool depths and volumes compared to pools in basins unaffected by ATV usage. These changes are logical consequences of accelerated erosion and sediment deposition, both of which have been associated with roads and trails in past research. Since the ATV trail system is the only important difference between these basins, we conclude that its presence and usage have created the observed differences. More work is needed to determine whether there are differences between designed and undesigned trails, and which specific features of ATV trails (e.g., slope and distance from a stream) are most responsible for these impacts. Results from these new studies will have important implications for ATV management and the maintenance of healthy stream ecosystems in the ONF.

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EFFECTS OF RETAINING A HARDWOOD COMPONENT DURING THE APPLICATION OF UNEVEN-AGED SILVICULTURE IN A SHORTLEAF PINE-OAK STAND: 6-YEAR RESULTS

Michael G. Shelton[1]

Abstract—Treatments were the following hardwood basal areas (square feet per acre) and spatial arrangements: 0, 15-grouped, 15-scattered, 30-scattered, and an untreated control. Pine basal area was reduced by harvesting to 60 square feet per acre in all treatments except the control. After six growing seasons, pine regeneration ranged from 8,890 stems per acre in the 0-hardwood treatment to 0 stems per acre in the control, and stems were 6.2 times taller in the 0-hardwood treatment than in the 30-scattered treatment. Height growth of pine regeneration was acceptable in the 0-hardwood and 15-grouped treatments, but was inadequate elsewhere. Oak regeneration averaged 1,510 stems per acre after 6 years with no significant differences among treatments. Coverage of understory vegetation was greatest in the 0-hardwood treatment and declined as hardwood retention increased. Volume growth of merchantable pines surviving the 6-year period was 34 percent higher in the 0-hardwood treatment than in the 15- and 30-scattered treatments. However, high rates of postharvest mortality resulted in no significant differences in rates of net growth among treatments.

INTRODUCTION

Knowledge about implementing uneven-aged silviculture in shortleaf pine (*Pinus echinata* Mill.) stands is very limited (Murphy and others 1991). Techniques and guidelines developed for loblolly (*P. taeda* L.)–shortleaf pine stands at the Crossett Experimental Forest in the Coastal Plain of Arkansas (Baker and others 1996) may be adaptable, but most of the Crossett experience applies to managing existing uneven-aged pine stands or rehabilitating understocked pine stands. Stands in the Ouachita Mountains are mostly mature pines and oaks (*Quercus* spp.) that have developed an even-aged character, and a hardwood component is desired to enhance nontimber resources. Such stands pose a unique set of problems: (1) some hardwoods must be removed to create a favorable environment for the establishment and development of pine regeneration; (2) a reverse-J diameter distribution must be developed, which will likely take several decades; and (3) the response of suppressed pines of the original stand to release is questionable (Shelton and Murphy 1991, 1993).

This ongoing research focuses on some of the problems of applying uneven-aged silviculture in mature pine-oak stands on poor sites. Such stands are common on public lands, and knowledge is needed concerning alternative silvicultural systems. Study objectives are to test the traditional application of uneven-aged pine silviculture using single-tree selection and to determine the limits for hardwood retention within this system. Response of understory vegetation and merchantable growth is reported at 6 years after implementation of the study. This paper updates an earlier one reporting 3-year results (Shelton and Murphy 1997).

METHODS

Study Area

The study was installed on the Winona Ranger District of the Ouachita National Forest in Perry County, AR. Plots were oriented along an east-west ridge, which is typical of the physiography of the Ouachita Mountains. Elevations ranged from 640 to 790 feet above sea level. Blocks were located on the following slope positions: the lower, middle, and upper north slopes and the upper south slope. Slopes of individual plots ranged from 8 to 21 percent; aspects ranged from north to northwest on the north-slope positions and from southeast to southwest on the south-slope position.

Soils of the study area are mapped as the Carnasaw and Pirum series, both Typic Hapludults. These are well-drained, moderately deep soils that developed in colluvium and residuum weathered from sandstone and shale. Natural fertility and organic matter are low, and the soils are strongly acidic. Site index for shortleaf pine averaged 57 feet at 50 years and ranged from 53 to 64 feet, which is typical of upland sites in the Ouachita Mountains (Graney 1992). The lower north slope had a slightly higher site index than the other three slope positions (61 versus 56 feet). Site index averaged 53 feet at 50 years for white oak (*Quercus alba* L.) and 54 feet for black oak (*Q. velutina* Lam.).

Vegetation in the study area was typical of forested landscapes in the Ouachita Mountains, where upland forests are dominated by shortleaf pine and mixed oaks (Guldin and others 1994). Before treatment, overstory basal area [trees ≥ 3.6 inches in diameter at breast height (d.b.h.)] in this mature, second-growth, shortleaf pine-oak stand averaged 90 square feet per acre for shortleaf pine and 32 square feet per acre for hardwoods. Oaks accounted for 84 percent of the total hardwood basal area. White oak was the most prevalent hardwood, with lesser amounts of post oak (*Q. stellata* Wangenh.), black oak, blackjack oak (*Q. marilandica* Muenchh.), and southern red oak (*Q. falcata* Michx.). The remaining 16 percent of hardwood basal area was ash (*Fraxinus* spp.), hickory (*Carya* spp.), red maple (*Acer rubrum* L.), serviceberry [*Amelanchier arborea* (Michx. f.) Fern.], blackgum (*Nyssa sylvatica* Marsh.), and flowering dogwood (*Cornus florida* L.). The understory was mainly

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regeneration of the more shade-tolerant species and a variety of common shrubs, such as huckleberries (*Vaccinium* spp.) and hawthorns (*Crataegus* spp.).

Overstory pines and oaks ranged in age from 30 to over 110 years (Shelton and Murphy 1991). However, most of the pines were 50 to 80 years old, and the oaks were 40 to 70 years old. The scarcity of younger overstory trees indicated that regeneration and subsequent recruitment of both the pines and oaks had been limited for 30 to 40 years of stand development.

Study Design and Treatments

Sixteen square 0.5-acre plots were installed and surrounded by a 58.2-foot isolation strip that received the same treatment. Basal area of overstory pines (trees ≥ 3.6 inches d.b.h.) was reduced to 60 square feet per acre in all plots. Treatments retained the following hardwood basal areas (square feet per acre) and spatial arrangements for overstory trees: 0, 15-grouped, 15-scattered, and 30-scattered. Openings in the pine canopy ranged from 0.10 to 0.25 acre and often extended from the 0.5-acre plots into the isolation strip. In the scattered arrangements, hardwoods were uniformly distributed across each plot, while in the grouped arrangement they were located outside openings in the pine canopy. For the grouped arrangement, no attempt was made to create openings in the pine canopy other than those resulting from application of single-tree selection. The grouped spatial arrangement was similar to the 0-hardwood treatment within openings and to the 30-scattered treatment outside openings. Treatments were assigned in a randomized, complete block design with four replications of each treatment. In 1991, permanent 0.5-acre plots were established in untreated areas adjacent to each block to serve as untreated controls.

The pine component was regulated using the basal area–maximum diameter–quotient (BDq) method of single-tree selection (Baker and others 1996). Targets were 60 square feet per acre for basal area, 18 inches for maximum d.b.h., and a quotient of 1.2 for 1-inch d.b.h. classes. Targets for maximum diameter and quotient were followed as closely as feasible, because the stand lacked a balanced reverse-J diameter distribution characteristic of uneven-aged structure. Hardwood retention favored the higher quality red and white oaks, which typically were the largest hardwoods in the study area.

Plots were harvested from December 1988 through early March 1989 using mules to skid logs to landings. Because there were no local markets for hardwoods, all hardwoods ≥ 1 inch d.b.h. that were not designated for retention were injected with triclopyr amine in April 1989. Herbicide treatments were applied by contract crews following label directions, although research crews did some followup work.

Measurements

During March 1989, all retained pines and hardwoods (≥ 3.6 inches d.b.h.) in the 0.5-acre plots were measured for d.b.h., and the location of each stem was mapped by determining azimuth and distance from plot center. About one-third of the trees were measured for height and age. Ten permanent points were systematically located within each 0.5-acre plot for monitoring the development of understory vegetation. The monitoring points were located so that none were closer than 30 feet from the 0.5-acre plot boundary and 88 feet from the isolation boundary.

In June 1994, coverage of understory vegetation was visually estimated in milacre plots (3.72-foot radius) centered around the 10 permanent monitoring points in each 0.5-acre plot. The following groups were used: grasses, forbs, vines, shrubs, hardwoods, pines, and total vegetation. Evaluations were conducted in June because coverage was observed to maximize before the summer droughts that typically occur on such sites. During mid-September 1994, all woody seedlings (≤ 0.5 inch d.b.h.) were counted in the milacre plots by species or species group. Multiple-stemmed rootstocks were tallied as one individual. Woody saplings (0.6 to 3.5 inches d.b.h.) were counted in 0.01-acre circular subplots centered around each of the 10 permanent points per 0.5-acre plot. The two tallest pines (if any) and two tallest hardwoods in each 0.01-acre subplot were selected as the dominant regeneration and were measured for groundline diameter, height, and crown width.

Data Analysis

Mean values for understory vegetation were calculated for the 10 regeneration subplots in each 0.5-acre plot. Regeneration subplots were considered to be stocked by pine and nonpine woody species if at least one individual was present to represent the species or species group. To facilitate data presentation, nonpine species were grouped as follows: oaks, other overstory trees, midcanopy trees, and shrubs. Other overstory trees included blackgum, hickory, ash, and sweetgum (*Liquidambar styraciflua* L.). Midcanopy trees included red maple, serviceberry, dogwood, elms (*Ulmus* spp.), persimmon (*Diospyros virginiana* L.), and black cherry (*Prunus serotina* Ehrh.). Shrubs were mostly huckleberries, hawthorns, and plums (*Prunus* spp.).

Pine volume in each 0.5-acre plot was calculated using taper curves for natural shortleaf pine (Farrar and Murphy 1987). Inside-bark cubic-foot volume for merchantable trees (d.b.h. ≥ 3.6 inches) was computed from a 1-foot stump to a 4.0-inch diameter outside-bark top. Volume for sawtimber trees (d.b.h. ≥ 9.6 inches) was computed from a 1-foot stump to an 8-inch diameter outside-bark top. Cubic-foot volume was inside bark. Hardwood volume was calculated using equations of Clark and others (1986). Hardwood merchantability limits were the same as for pines except that stump heights varied as follows: 0.2 foot for trees 3.6 to 4.9 inches d.b.h., 0.6 foot for trees 5.0 to 10.9 inches d.b.h., and 1.0 foot for larger trees. Sawtimber volume was not calculated for hardwoods because of their small size and generally poor quality.

Analysis of variance for a randomized, complete block design was used to compare expressions of understory vegetation and volume growth among treatments. Significance was accepted at a probability level (P) ≤ 0.05. Density and stocking values were transformed to homogenize the error term. Differences among treatment means were isolated using the Ryan-Einot-Gabriel-Welsch multiple range test ($P = 0.05$). This procedure, which is one of the most powerful step-down, multiple-range tests available, controls the experiment-wise error rate (SAS Institute 1989).

RESULTS AND DISCUSSION

Regeneration Density

Most shortleaf pine seedlings became established from seeds dispersed following harvest and site preparation. At 6 years, there was an average of 6,175 seedlings per acre in treatments with timber harvesting, but no significant differences occurred among these treatments (table 1). However, no pine seedlings were found in the untreated controls. Apparently, environmental conditions in the 30-scattered treatment were sufficiently different from the untreated control to allow the establishment of 6,075 seedlings per acre. The 30-scattered treatment had about the same merchantable hardwood basal area as the control, but the control had an additional 30 square feet per acre of pine basal area and 4 square feet per acre of submerchantable hardwoods. Shelton and Murphy (1997) reported a mean across all harvested treatments of only 756 seedlings per acre at 3 years, which means that most of the pine seedlings present at 6 years became established after 3 years.

The generally accepted minimum stocking for pine regeneration in uneven-aged pine stands is 200 submerchantable stems per acre (Cain and others 1987). Although all of the harvested treatments exceeded this minimum, shortleaf pine seedlings in the 15-scattered and 30-scattered treatments are not expected to survive. Other researchers have observed that pine seedlings can become established under a dense canopy and persist for several years before dying (Becton 1936, Wahlenberg 1960). This observation suggests that newly established pine seedlings are moderately shade tolerant but become shade intolerant with age. Bormann (1956) reported, for example, that the photosynthetic efficiency of loblolly pine seedlings at low light intensities declined substantially as secondary growth characteristics developed.

Shortleaf pine saplings occurred only in the 0-hardwood (65 stems per acre) and 15-grouped (15 stems per acre) treatments at 6 years, and differences among treatments were significant. Saplings in the 15-grouped treatment were only in openings where no hardwoods were retained.

For nonpine species, an average of 8,365 rootstocks per acre occurred in the seedling size class at 6 years with no significant treatment differences (table 1). This represents a 40-percent decline from levels reported at 3 years (Shelton and Murphy 1997) and probably reflects mortality from self-thinning and outgrowth to the sapling size class. Shrubs were the most common nonpine species group, and oaks represented 17 percent of the total.

Treatments significantly affected the density of sapling oaks and other canopy trees; the fewest saplings occurred in the 30-scattered treatment where they had been controlled during site preparation. The greatest density of oak saplings occurred in the 0-hardwood treatment (128 stems per acre) and the untreated control (132 stems per acre). Saplings of other canopy species were most prevalent in the 0-hardwood treatment (108 stems per acre) followed by the 15-grouped

Table 1—Density of seedlings and saplings 6 years after the initial harvest implementing uneven-aged silviculture in a shortleaf pine-hardwood stand

	Density (number/acre) by hardwood treatment						
Species or group	0 ft^2/ac	15 ft^2/ac grouped	15 ft^2/ac scattered	30 ft^2/ac scattered	Control	Mean square error[a]	*P*[b]
			seedlings				
Shortleaf pine	8,825a[c]	3,275a	6,525a	6,075a	0b	571	<0.01
Oaks	825	1,225	1,900	1,100	2,175	135	0.22
Other canopy trees	1,375a	925ab	925ab	1,300a	100b	111	0.02
Midcanopy trees	350	1,825	1,100	1,300	700	272	0.50
Shrubs	2,275	3,050	4,550	6,425	8,400	1,031	0.37
Nonpine total	4,825	7,025	8,475	10,125	11,375	349	0.12
			saplings				
Shortleaf pine	65a	15ab	0b	0b	0b	6.1	<0.01
Oaks	128a	35b	28bc	2c	132a	3.1	<0.01
Other canopy trees	108a	72a	28ab	5b	65a	5.6	<0.01
Midcanopy trees	55	55	10	5	60	13.2	0.32
Shrubs	10	2	0	2	12	3.8	0.42
Nonpine total	301a	164ab	66bc	14c	269a	7.2	<0.01

[a] Based on transformed values: (number per acre)$^{0.5}$.
[b] Probability level.
[c] Row means followed by the same letter or no letter are not significantly different at $P = 0.05$.

treatment (72 stems per acre). Most saplings in the 15-grouped treatment were located in the openings created by hardwood control.

Regeneration Stocking

Percentage stocking is often a better measure of regeneration success than density because stocking expresses the spatial distribution of seedlings, whereas density does not. Stocking is less sensitive to the clumped-spatial pattern that often occurs with natural regeneration (Daniels 1978). The stocking of shortleaf pine seedlings and saplings was similar in pattern to density. For the harvested treatments, pine seedling stocking averaged 66 percent with no significant differences (table 2). The generally accepted optimum stocking limit for pine regeneration in uneven-aged pine stands is 50 percent (Cain and others 1987). All harvested treatments either met or exceeded this limit at 6 years, although seedlings are not expected to survive in the 15-scattered and 30-scattered treatments. The treatments had a significant effect on the stocking of shortleaf pine saplings, which occurred only in the 0-hardwood and 15-grouped treatments.

A nonpine species in the seedling size class occurred on nearly all milacre subplots; stocking levels averaged 92 percent with no significant differences among treatments. Stocking for oaks averaged 58 percent with no significant differences among treatments. Seedlings of other canopy trees had a significantly lower stocking in the unharvested control (8 percent) than in the harvested treatments (47 percent).

For stocking of nonpine saplings, significant differences occurred among treatments for oaks, other canopy trees, and all nonpines. The pattern displayed across treatments was the same for each species group; the 0-hardwood treatment nearly equaled the control, and stocking decreased in the following order: 0-hardwood > 15-grouped > 15-scattered > 30-scattered.

Size of Dominant Regeneration

After the sixth growing season, shortleaf pine regeneration decreased in size as hardwood retention increased (table 3). Pines in the 0-hardwood treatment were larger than those in the 30-scattered treatment by 7.5 times for groundline diameter, 4.1 times for crown width, and 6.2 times for height. Differences were highly significant in all cases. Baker and others (1996) state that annual height growth of pines should be ≥ 0.5 feet for acceptable development. This guideline was met in the 0-hardwood and 15-grouped treatment but not in the other treatments.

The nonpine groups showed a pattern similar to the pines except that the magnitude of suppression was not as great. For example, regeneration in the 0-hardwood treatment was taller than in the 30-scattered treatment by 2.7 times for the oaks, 1.9 for other canopy trees, and 1.9 for midcanopy trees. Hardwood regeneration was still larger than shortleaf pine regeneration at 6 years. This reflected a difference in the principal reproductive strategy of the two groups—seeds for pines versus advanced regeneration and sprouts for hardwoods. Experience elsewhere has shown that the height

Table 2—Stocking of seedlings and saplings 6 years after the initial harvest implementing uneven-aged silviculture in a shortleaf pine-hardwood stand

Species or group	Stocking (percent) by hardwood treatment[a] 0 ft²/ac	15 ft²/ac grouped	15 ft²/ac scattered	30 ft²/ac scattered	Control	Mean square error[b]	P[c]
			seedlings				
Shortleaf pine	78a[d]	60a	65a	62a	0b	0.062	<0.01
Oaks	50	50	70	52	70	0.041	0.31
Other canopy trees	52a	50a	42a	45a	8b	0.025	<0.01
Midcanopy trees	25	48	45	45	32	0.080	0.58
Shrubs	38	52	52	58	60	0.061	0.67
All nonpines	88	92	98	92	92	0.062	0.76
			saplings				
Shortleaf pine	25a	10ab	0b	0b	0b	0.024	<0.01
Oaks	58a	22b	15bc	2c	58a	0.021	<0.01
Other canopy trees	50a	40a	18ab	5b	45a	0.053	<0.01
Midcanopy trees	28	25	8	5	25	0.076	0.45
Shrubs	8	2	0	2	10	0.034	0.43
All nonpines	88a	58ab	35bc	15c	88a	0.047	<0.01

[a] Stocking is based on milacre plots for seedlings and 0.01-acre plots for saplings.
[b] Based on transformed values: arcsine $(\text{percent}/100)^{0.5}$.
[c] Probability level.
[d] Row means followed by the same letter or no letter are not significantly different at $P = 0.05$.

Table 3—Mean size of the dominant regeneration 6 years after the initial harvest implementing uneven-aged silviculture in a shortleaf pine-hardwood stand

	Regeneration size by hardwood treatment[a]					
Species or group	0 ft^2/ac	15 ft^2/ac grouped	15 ft^2/ac scattered	30 ft^2/ac scattered	Mean square error	*P*[b]
	- - - - *Groundline diameter* (*inches*) - - - -					
Shortleaf pine	0.82a[c]	0.53ab	0.24bc	0.11c	0.033	<0.01
Oaks	1.71a	0.87b	0.91b	0.66b	0.093	<0.01
Other canopy trees	1.50a	1.21ab	1.08bc	0.80c	0.036	<0.01
Midcanopy trees	1.50a	1.21ab	0.83b	0.76b	0.045	0.03
	- - - - - - *Crown diameter* (*feet*) - - - - - - -					
Shortleaf pine	1.96a	1.48a	0.80b	0.48b	0.104	<0.01
Oaks	5.11a	3.17b	3.33b	2.69b	0.503	<0.01
Other canopy trees	5.45a	4.28ab	4.27ab	3.98b	0.357	0.03
Midcanopy trees	5.72	5.61	3.72	3.62	1.184	0.10
	- - - - - - - - *Total height* (*feet*) - - - - - - - -					
Shortleaf pine	4.66a	3.19ab	1.61bc	0.75c	0.89	<0.01
Oaks	9.07a	5.05b	5.17b	3.41b	2.28	<0.01
Other canopy trees	9.40a	7.34b	7.46b	4.93c	0.90	<0.01
Midcanopy trees	11.47a	9.44ab	6.74b	5.95b	2.07	0.04

[a] The unharvested control was not included in this comparison because no pines occurred and the hardwoods present were a different age class than those in the harvested treatments.
[b] Probability level.
[c] Row means followed by the same letter or no letter are not significantly different at $P = 0.05$.

growth of free-to-grow pine regeneration will eventually exceed that of hardwoods on most upland sites (Wahlenberg 1960), but the 6-year results of this study may be too early to confirm this. At 3 years, hardwoods were 2.1 times taller than pines in the 0-hardwood treatment (Shelton and Murphy 1997), and this was the same value observed at 6 years.

Understory Coverage

The effect of treatments on understory coverage varied among species groups, with significant differences occurring for grasses, vines, shortleaf pine, and total vegetation (table 4). Generally, coverage declined as more overstory hardwoods were retained, although some variation occurred in this pattern. Coverage was greater in the 0-hardwood treatment than in the control by 15.1 times for grasses, 2.4 for vines, and 2.1 for hardwoods. In contrast, shrub coverage was higher in the unharvested control than in the harvested treatments. Comparison of the 6-year results with those reported at 3 years (Shelton and Murphy 1997) showed that pines and hardwoods doubled in coverage, vines increased slightly, and grasses decreased. This suggests that pines and hardwoods will dominate the subsequent pattern of successional change in understory vegetation.

Growth of Merchantable Trees

Postharvest mortality of shortleaf pines was high and required expressing growth by its functional components—survivor growth, mortality, and net growth (Husch and others 1982). Survivor growth for total merchantable volume was significantly higher in the 0-hardwood treatment (42 cubic feet per acre per year) than in the 15- and 30-scattered treatments by 34 percent (table 5). Mortality losses of shortleaf pine averaged 9 cubic feet per acre per year for merchantable volume with no significant differences among treatments; mortality averaged 1.0 tree per acre per year and was mostly from the smaller trees in the stand. Dying trees had a mean d.b.h. of 7.4 inches compared with 10.1 inches for surviving trees. Tree mortality was from multiple causes, such as insects, lightning, suppression, and windthrow, but in the majority of cases the cause was not discernable (58 percent). Cain (1998) also observed high rates of inexplicable mortality following the implementation of uneven-aged silviculture in a shortleaf pine stand in southern Arkansas. Net growth of merchantable pines averaged 25 cubic feet per acre per year with no significant differences among treatments.

Net volume growth for sawtimber averaged 122 board feet (Doyle) per acre per year for shortleaf pine with no significant differences among treatments (table 5). Murphy and others (1991) reported sawtimber growth in three well-structured, uneven-aged shortleaf pine stands in the Ouachita Mountains. In a stand with the same site index as this study (57 feet at 50 years), they reported growth of 209 board feet (Doyle) per acre per year; on poorer sites they reported lower growth rates (137 and 126 board feet per acre per year for stands with site indices of 50 and 55 feet, respectively). Thus, the sawtimber growth for this study, which lacks an uneven-aged structure, appears somewhat lower than well-structured stands on similar sites.

Table 4—Coverage of understory vegetation 6 years after the initial harvest implementing uneven-aged silviculture in a shortleaf pine-hardwood stand

	Understory coverage by hardwood treatment						
Species or group	0 ft^2/ac	15 ft^2/ac grouped	15 ft^2/ac scattered	30 ft^2/ac scattered	Control	Mean square error[a]	P[b]
	- - - - - - - - - - *percent* - - - - - - - - - -						
Grasses	24.1a[c]	8.4bc	10.8b	4.5cd	1.6d	0.003	<0.01
Forbs	1.8	1.5	0.9	1.4	0.2	0.006	0.59
Vines	22.0ab	27.0a	16.0ab	7.7b	9.0b	0.012	0.03
Shrubs	7.6	6.5	5.5	7.0	10.9	0.016	0.72
Hardwoods	31.9	29.5	16.1	13.2	15.1	0.016	0.07
Shortleaf pine	3.9a	1.1ab	0.3b	0.2b	0.0b	0.003	<0.01
Total vegetation[d]	78.4a	63.8ab	45.5bc	33.9c	32.0c	0.010	<0.01

[a] Based on transformed values: arcsine (percent/100)$^{0.5}$.

[b] Probability level.

[c] Row means followed by the same letter or no letter are not significantly different at $P = 0.05$.

[d] Coverage of total vegetation is less than the sum of individual species groups because of multiple occupancy.

Table 5—Growth of merchantable-sized trees for the 6-year period after the initial harvest implementing uneven-aged silviculture in a shortleaf pine-hardwood stand

	Annual growth by hardwood treatment[a]					
Growth component by species or group	0 ft^2/ac	15 ft^2/ac grouped	15 ft^2/ac scattered	30 ft^2/ac scattered	Mean square error	P[b]
	- - - - *Merchantable volume (ft^3/ac)* - - - -					
Shortleaf pine						
Survivor	42.3a[c]	33.4ab	32.3b	30.6b	24	0.03
Mortality	17.6	7.3	3.1	9.8	269	0.66
Net	24.7	26.1	29.2	20.8	388	0.94
Hardwoods						
Survivor	—	12.4b	16.0ab	20.0a	19	0.12
Mortality	—	2.7	0.4	1.4	9	0.58
Net	—	9.7b	15.6ab	18.6a	17	0.05
	- - - - - *Sawtimber volume*[d] (*ft^3/ac*) - - - - -					
Shortleaf pine						
Survivor	39.2	38.9	33.7	30.8	41	0.24
Mortality	14.0	4.9	0.0	8.5	248	0.65
Net	25.2	34.0	33.7	22.3	336	0.74
	- - *Sawtimber volume*[d] (*fbm Doyle/ac*) - -					
Shortleaf pine						
Survivor	166	157	133	132	688	0.24
Mortality	56	15	0	31	3,976	0.65
Net	110	142	133	101	5,148	0.83

[a] Control plots were not monitored for growth.

[b] Probability level.

[c] Row means followed by the same letter or no letter are not significantly different at P=0.05.

[d] Too few hardwoods occurred in sawtimber size classes for analysis.

Hardwoods contributed to the stand's growth in total merchantable volume; net growth averaged 15 cubic feet per acre per year. Mortality of hardwoods was considerably less than shortleaf pine and averaged only 1.5 cubic feet per acre per year. Because of their small size and generally low quality, hardwoods did not contribute to the stand's sawtimber growth.

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

The amount and spatial distribution of natural pine regeneration depend on seed supply, seedbed conditions, and environmental conditions. The treatments evaluated in this study principally affected the environmental conditions that govern seedling survival and growth. Because of shortleaf pine's shade intolerance, seedling development will be suppressed to some extent in any reproductive cutting method that retains an overstory. The key to successful application of uneven-aged silviculture is to reach a balance between retaining an adequate overstory stocking for acceptable merchantable growth and reducing overstory stocking enough to provide the environmental conditions needed for regeneration. Pine growth rates in both the overstory and understory will be below their short-term potential, but this compromise provides a system that should be sustainable through time.

Recurring pine regeneration is crucial for the long-term sustainability of uneven-aged stands in which shortleaf pine is the dominant species. Retaining hardwoods within the single-tree selection system will most severely impact environmental conditions in the understory. Hardwoods apparently suppress the development of pine seedlings more than an equivalent basal area of pines, which reflects differences in crown and foliar features. Thus, acceptable limits for hardwoods within an uneven-aged pine stand are apparently low, even if the pine basal area is reduced accordingly. Early results of this study show that retaining as little as 15 square feet per acre of hardwood basal area in a scattered distribution with a pine basal area of 60 square feet per acre (the 15-scattered treatment) will prevent the acceptable development of sufficient pine regeneration to sustain long-term pine timber production. However, reducing pine basal area to 45 square feet per acre might allow retention of 5 to 10 square feet per acre of scattered hardwoods (Baker and others 1996).

Results also suggest that 15 square feet per acre of hardwood basal area can be retained outside openings in the pine canopy (the 15-grouped treatment) while still obtaining adequate development of pine regeneration within the openings. However, the success of this treatment can only be measured by evaluating long-term developmental rates of regeneration. In addition, the feasibility of implementing the 15-grouped treatment needs to be operationally tested. This treatment was difficult to implement even on these small research plots, because it was hard to determine where opening boundaries occurred. Creating openings to promote stand regeneration would have been easier using group selection, where both pines and hardwoods are removed within well-defined openings (Murphy and others 1993).

This study shows that adequate pine regeneration will occur in shortleaf pine stands in the Ouachita Mountains when: (1) traditional guidelines for uneven-aged pine silviculture are followed (that is, the 0-hardwood treatment), (2) seed supply and environmental conditions are favorable, and (3) competing understory vegetation is low. More time, however, is needed to fully assess developmental rates of regeneration and to evaluate the growth of overstory trees. Although traditional guidelines for uneven-aged pine stands exclude a hardwood component at a local scale, they can be retained at the stand level along drainages or in clumps or clusters in an area-wise distribution. Permissible levels of hardwood retention within this spatial pattern are probably not limited by biology, as long as pine-dominated areas are large enough to provide relative freedom from the edge effects of retained hardwoods. Rather, landowner objectives and operational concerns will probably set realistic limits. According to Baker and others (1996), the area-wise distribution of pines and hardwoods in uneven-aged stands under single-tree selection has a number of advantages for landowners who want to retain hardwoods: (1) a favorable environment is provided for the shade-intolerant pine regeneration, (2) a significant hardwood component can be retained, (3) species-site relationships are optimized, (4) sensitive areas are protected, (5) hardwoods are protected during silvicultural operations, (6) stand regulation and marking are simplified, and (7) a varied wildlife habitat is provided.

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UNDERSTORY VEGETATION AND OVERSTORY GROWTH IN PINE AND PINE-HARDWOOD SHELTERWOOD STANDS IN THE OUACHITA MOUNTAINS: 5-YEAR RESULTS

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Abstract—Treatments were two overstory compositions (a pine basal area of 30 square feet per acre with and without 15 square feet per acre of hardwoods) and two methods of submerchantable hardwood control (chainsaw felling with and without stump-applied herbicide). After the fifth growing season, pine regeneration averaged 1,870 seedlings per acre and 500 saplings per acre in the pine-overstory treatment; there were more pine seedlings when overstory hardwoods were retained (3,090 seedlings per acre) but fewer saplings (27 saplings per acre). Pine regeneration was twice as tall in the pine-overstory treatment than in the pine-hardwood treatment. Oak regeneration averaged 1,100 stems per acre. Overstory pines responded to hardwood removal, averaging 0.84 square feet per acre per year for basal area growth in the pine-overstory treatment and 0.58 square feet per acre per year in the pine-hardwood treatment. Very few overstory pines died after harvesting (0.04 trees per acre per year). Results indicate that 15 square feet per acre of scattered hardwoods can be retained through at least 5 years after harvest.

INTRODUCTION

The shelterwood reproduction cutting method is a versatile way of naturally regenerating even-aged or two-aged stands that is increasingly being used on national forest lands. This method gradually removes mature trees in a series of partial cuts and retains more trees than other even-aged reproduction cutting methods. The higher density of retained trees may satisfy some silvicultural and landowner objectives by making the stand more visually pleasing during regeneration and increasing timber yields through enhanced growth of high-quality trees. Shelterwood stands also tend to have high rates of seed production, which may improve chances for successful natural regeneration in areas with low or erratic seed production. For example, Brender and McNab (1972) reported that enough seeds for successful regeneration were produced 50 percent of the time in shelterwood stands compared with 21 percent in seed-tree stands. The shelterwood method may be favorable in regenerating shortleaf pine (*Pinus echinata* Mill.) stands in the Ouachita Mountains, where seed production is variable (Shelton and Wittwer 1996) and may reduce the intensity of site preparation needed.

Hardwood retention is desired to enhance nontimber resources on many public and nonindustrial private lands, and a pine-hardwood mixture is often the target composition in the regenerated stand. Objectives of this study are to test the traditional application of the shelterwood reproduction cutting method in shortleaf pine-oak stands, to evaluate the effects of hardwood retention within this system, and to determine the type of submerchantable hardwood control needed. The development of understory vegetation is reported 5 years after harvesting and hardwood control, and this paper updates an earlier one reporting 3-year results (Shelton 1997).

METHODS

Study Area

The study area was located in the Winona Ranger District of the Ouachita National Forest in Perry County, AR. Plots were oriented along an east-west ridge typical of the physiography of the Ouachita Mountains. Elevations ranged from 640 to 810 feet above sea level. Blocks were located along the lower, middle, and upper north slope and the upper south slope. Slopes of individual subplots ranged from 5 to 26 percent. Aspect was south on the south-slope position and ranged from north to east on the north-slope positions.

Soils of the study area are mapped as the Carnasaw and Pirum series, both Typic Hapludults. These are well drained, moderately deep soils that developed in colluvium and residuum weathered from sandstone and shale. Natural fertility and organic matter are low, and the soils are strongly acidic. Site index for shortleaf pine averaged 60 feet at 50 years, ranging from 56 to 65 feet; this is typical of upland sites in the Ouachita Mountains (Graney 1992). The lower north slope had a slightly higher site index than the other slope positions (62 versus 59 feet). White oak (*Quercus alba* L.) site index averaged 54 feet at 50 years. The dominant pines were slightly older than the dominant white oaks (66 versus 61 years).

Vegetation in the study area was typical of forested landscapes in the Ouachita Mountains, where mature, second-growth shortleaf pine and mixed oaks dominate upland forests (Guldin and others 1994). Preharvest overstory basal area in trees ≥ 3.6 inches in diameter at breast height (d.b.h.) averaged 74 square feet per acre for shortleaf pine and 41 square feet per acre for hardwoods. Oaks accounted for 92 percent of the hardwood basal area. White oak was the most prevalent hardwood, followed by post oak (*Q. stellata* Wangenh.), black oak (*Q. velutina* Lam.), blackjack oak (*Q. marilandica* Muench.), and southern red oak (*Q.*

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falcata Michx.). The remaining 8 percent of hardwood basal area was ash (*Fraxinus* spp.), hickory (*Carya* spp.), red maple (*Acer rubrum* L.), serviceberry [*Amelanchier arborea* (Michx. f.) Fern.], blackgum (*Nyssa sylvatica* Marsh.), and dogwood (*Cornus florida* L.). The understory consisted of tree regeneration, mainly the more shade-tolerant species, and a variety of common shrubs, such as huckleberries (*Vaccinium* spp.), snowbell (*Styrax grandifolia* Ait.), and hawthorns (*Crataegus* spp.).

Study Design and Treatments

The study was a split-plot design with four randomized complete blocks. Two overstory compositions (pine only and pine-hardwood) were established in eight 3.50-acre whole plots, each split into two 1.75-acre subplots for testing two hardwood control methods (manual versus chemical control of submerchantable hardwoods). Each subplot consisted of a 0.70-acre measurement area (103 by 295 feet) and an isolation strip of 1.05-acre. The isolation strip for a subplot was 66 feet wide when adjacent to the untreated stand, 50 feet wide when adjacent to the neighboring whole plot that made up the block, and 20 feet wide along the internal boundary separating the two subplots within a whole plot. Within each 0.70-acre measurement area, 18 permanent points were systematically located to monitor understory vegetation. These points were at least 80 feet from the external boundary of the whole plot and 50 feet from the internal boundary separating the two subplots.

Target retention for all overstories was 30 square feet per acre of pine basal area. The pine-overstory treatment had no hardwoods; retention in the pine-hardwood treatment was 15 square feet per acre of hardwoods and 30 square feet per acre of pines. Pine seed trees were selected for a past history of high cone production, a d.b.h. of 12 to 16 inches, good vigor and stem quality, and a uniform spatial distribution. Selection criteria were sometimes relaxed to achieve the target basal area. Large, well-formed, vigorous red and white oaks were preferred for the hardwood component, but less desirable species or low-quality stems were sometimes kept to meet target basal area.

Whole plots were temporarily subdivided into seven 0.5-acre areas to facilitate marking the target basal areas for overstory trees and to ensure a uniform distribution. All merchantable pines, except seed trees, were harvested using rubber-tired skidders and tree-length skidding from November 1989 to early January 1990. A commercial firewood vendor harvested merchantable hardwoods ≥ 6 inches d.b.h. beginning in early April 1990 but was stopped shortly afterward by wet weather. Harvesting resumed in July and was completed in October.

Treatments to control submerchantable hardwoods (0.6 to 5.5 inches d.b.h.) began in mid-August 1990 and were completed in early September 1990. Treatments were either manual (chainsaw felling) or chemical (chainsaw felling followed immediately by application of undiluted Garlon® 3A to the stump). In the manual treatment, submerchantable pines were chainsaw felled but not treated with herbicide.

Measurements

Understory vegetation was inventoried during mid-September 1995 using 18 permanent plots for each 0.70-acre subplot. Horizontal coverage of understory vegetation (grasses, herbs, vines, shrubs, hardwoods, pines, and total vegetation) was estimated on milacre plots centered on each point. All woody plants in the seedling size class (≤ 0.5 inch d.b.h.) were counted on the milacre plot, tallying multiple-stemmed rootstocks as one individual. Woody saplings (0.6 to 3.5 inches d.b.h.) were counted by species or species group and 1-inch d.b.h. classes on a 0.01-acre plot centered around each permanent point. On each 0.01-acre plot, the tallest two pines (if present) and tallest two hardwoods were measured for groundline diameter, crown width, and total height.

In March 1990, all retained pines and hardwoods (≥ 3.6 inches d.b.h.) on the 0.7-acre subplots were measured for d.b.h., and stem location was mapped by determining azimuth and distance from plot center. All pines and about a third of the hardwoods were measured for height. Tree d.b.h. was reinventoried biennially during the dormant season.

Data Analysis

Milacre plots were considered stocked by pine or deciduous woody species if at least one seedling was present for the species or species group; similarly, 0.01-acre plots were considered stocked if at least one sapling was present. Means were calculated for understory variables for each 0.7-acre subplot. To facilitate data presentation, deciduous species were grouped as oaks, other canopy trees, midcanopy trees, or shrubs. Other canopy trees included blackgum, hickory, ash, and sweetgum (*Liquidambar styraciflua* L.); midcanopy trees included maple, serviceberry, dogwood, elms (*Ulmus* spp.), persimmon (*Diospyros virginiana* L.), and black cherry (*Prunus serotina* Ehrh.). Shrubs included huckleberries, hawthorns, plums (*Prunus* spp.), snowbell, and several other common species.

Pine volumes were calculated from taper curves for natural shortleaf pine (Farrar and Murphy 1987). Inside-bark, cubic-foot volume for merchantable trees (d.b.h. ≥ 3.6 inches) was computed from a 1-foot stump to a 4.0-inch outside-bark top. Volumes for sawtimber trees (d.b.h. ≥ 9.6 inches) were computed from a 1-foot stump to an 8-inch, outside-bark top; cubic-foot volume was inside bark. Hardwood volumes were calculated from the equations of Clark and others (1986). Merchantability limits were the same as for pines except that stump heights varied as follows: 0.2 foot for trees 3.6 to 4.9 inches d.b.h., 0.6 foot for trees 5.0 to 10.9 inches d.b.h., and 1.0 foot for larger trees. Sawtimber volumes were not calculated for hardwoods because of their small size and generally poor quality.

Data were analyzed by analysis of variance for a split-plot randomized complete block design using the SAS procedure GLM (SAS Institute 1989). Since there were only two levels for each treatment, means were not separated but were presented with the associated mean square error (MSE) and probability level (*P*). Significance was accepted at $P \leq 0.05$.

Table 1—Density of seedlings and saplings in shelterwood stands with pine and pine-hardwood overstory compositions and manual and chemical methods of submerchantable-hardwood control

	Pine overstory		Pine-hardwood overstory		Overstory treatment		Hardwood control treatment	
Species or group	Manual	Chemical	Manual	Chemical	MSE[a]	*P*[b]	MSE	*P*
	- - - - - - - *Seedlings* (*number/acre*) - - - - - - -							
Shortleaf pine	1,500	2,236	2,764	3,416	6.47E5	0.06	4.51E6	0.54
Oaks	736	667	806	1,014	6.93E4	0.21	8.27E4	0.65
Other canopy trees	639	806	694	819	9.28E4	0.84	6.81E4	0.31
Midcanopy trees	444	944	500	514	2.14E5	0.48	2.10E5	0.31
Shrubs	3,680	3,806	4,792	5,194	4.32E6	0.32	5.70E6	0.83
Nonpine total	5,499	6,223	6,792	7,541	4.16E6	0.29	8.58E6	0.63
	- - - - - - - *Saplings* (*number/acre*) - - - - - - -							
Shortleaf pine	475	514	24	30	5.18E4	0.03	9.23E3	0.65
Oaks	476	374	179	163	1.31E4	0.02	6.13E3	0.18
Other canopy trees	174	182	89	71	4.95E3	0.07	1.28E4	0.94
Midcanopy trees	178	160	178	71	7.19E2	0.05	8.55E3	0.23
Shrubs	24	17	25	21	3.37E2	0.78	2.46E2	0.49
Nonpine total	852	733	471	326	6.20E5	<0.01	2.18E4	0.12

[a] Mean square error (MSE) are in exponential format; for example, 1.23E4 = 1.23X10^4 = 12,300.

[b] Probability level (*P*).

RESULTS AND DISCUSSION

Regeneration Density and Stocking

Virtually all the shortleaf pine seedlings became established from seeds dispersed after harvesting and site preparation. At 5 years, there was an average of 2,479 seedlings per acre with no significant differences between overstory or hardwood control treatments (table 1). However, hardwoods have been shown to substantially reduce the establishment of shortleaf pine regeneration when levels were greater than those tested in this study (Becton 1936, Shelton and Murphy 1997). Shelton (1997) reported an average of only 1,550 seedlings per acre at 3 years, so some shortleaf pine seedling establishment occurred between 3 and 5 years. The overstory treatment had a substantial and significant effect on the density of shortleaf pine saplings, averaging 494 stems per acre in the pine-overstory treatment and 27 stems per acre in the pine-hardwood treatment. By contrast, the submerchantable control treatment did not have a significant effect on sapling density. The generally accepted minimum density for pine regeneration in natural, even-aged pine stands is 700 stems per acre (Campbell and Mann 1973, Grano 1967). All overstory and hardwood control treatments either met or exceeded this at 5 years.

In the seedling size class, an average of 6,514 rootstocks per acre occurred for nonpine species at 5 years after harvesting with no significant treatment differences (table 1). This represented a 24-percent decline from levels reported at 3 years (Shelton 1997), which probably reflected mortality through self-thinning and outgrowth to the sapling size class. Shrubs were the most common nonpine species group, and the oaks represented 12 percent of the total. The overstory treatment significantly affected the sapling density of oaks and midcanopy trees, but the differences between treatments were not as great as with the pines. There were 2.5 times more oak saplings in the pine-overstory treatment than in the pine-hardwood treatment, but the treatment difference was only 1.4 times for midcanopy trees.

The stocking of shortleaf pine seedlings and saplings reflected a similar pattern as density. Stocking averaged 60 percent for seedlings, and no significant treatment differences occurred (table 2). The generally accepted minimum stocking for pine regeneration in natural, even-aged pine stands is 40 percent (Campbell and Mann 1973, Grano 1967). All overstory and hardwood control treatments either met or exceeded this at 5 years. The overstory treatment had a large and significant effect on sapling stocking; the pine-overstory treatment averaged 6.8 times more stocking than the pine-hardwood treatment. The submerchantable hardwood control treatment had no significant effect. A nonpine species occurred on nearly all regeneration plots, with stocking levels averaging 95 percent for seedlings and 90 percent for saplings. For saplings, oaks were the only group significantly affected by overstory treatment; stocking averaged 86 percent for the pine-overstory treatment compared with 55 percent for the pine-hardwood treatment.

Regeneration Size

After the fifth growing season, pine regeneration in the pine-overstory treatment was more than twice as large as that in the pine-hardwood treatment for groundline diameter, crown width, and height (table 3). Differences were highly significant in all cases. However, the hardwood control treatments did not significantly affect the size of pine regeneration. Oak regeneration responded to the overstory treatments in a manner similar to shortleaf pine, although the magnitude of the response was not as great. Differences were significant

Table 2—Stocking of seedlings and saplings in shelterwood stands with pine and pine-hardwood overstory compositions and manual and chemical methods of submerchantable-hardwood control

	Pine overstory		Pine-hardwood overstory		Overstory treatment		Hardwood control treatment	
Species or group	Manual	Chemical	Manual	Chemical	MSE[a]	P[b]	MSE	P
	- - - - - - - - *Seedlings*[c] (*percent*) - - - - - - - -							
Shortleaf pine	48	60	68	65	197	0.16	212	0.54
Oaks	43	40	46	57	90	0.13	225	0.60
Other canopy trees	29	44	30	42	227	0.91	218	0.13
Midcanopy trees	29	43	32	24	416	0.48	175	0.69
Shrubs	54	56	67	61	28	0.04	248	0.81
Nonpine total	87	94	93	92	61	0.75	92	0.57
	- - - - - - - - - *Saplings* (*percent*) - - - - - - - -							
Shortleaf pine	86	76	7	17	292	<0.01	87	0.98
Oaks	82	89	58	52	47	<0.01	120	0.90
Other canopy trees	48	58	43	38	348	0.25	206	0.75
Midcanopy trees	39	44	42	32	313	0.64	225	0.79
Shrubs	20	14	18	8	138	0.57	98	0.17
Nonpine total	96	97	88	80	88	0.07	73	0.53

[a] Mean square error (MSE).
[b] Probability level (*P*).
[c] Stocking is based on milacre plots seedlings and 0.01-acre plots for saplings.

Table 3—Size of the dominant regeneration in shelterwood stands with pine and pine-hardwood overstory compositions and manual and chemical methods of submerchantable-hardwood control

	Pine overstory		Pine-hardwood overstory		Overstory treatment		Hardwood control treatment	
Species or group	Manual	Chemical	Manual	Chemical	MSE[a]	P[b]	MSE	P
	- - - - - *Groundline diameter* (*inches*) - - - - -							
Shortleaf pine	1.8	1.8	0.7	0.8	0.08	<0.01	0.01	0.31
Oaks	2.6	2.5	1.9	1.8	0.03	<0.01	0.05	0.68
Other canopy trees	2.2	1.8	1.6	1.7	0.10	0.08	0.20	0.42
Midcanopy trees	1.7	1.8	1.7	1.3	0.02	0.02	0.30	0.95
	- - - - - - - - - *Crown width* (*feet*) - - - - - - - - -							
Shortleaf pine	3.2	3.3	1.5	1.7	0.17	<0.01	0.03	0.19
Oaks	6.6	6.4	5.0	5.1	0.22	<0.01	0.49	0.83
Other canopy trees	6.0	5.7	5.7	5.7	0.27	0.22	3.06	0.72
Midcanopy trees	6.1	6.4	5.5	4.9	0.82	0.06	3.43	0.86
	- - - - - - - - - - - *height* (*feet*) - - - - - - - - - - -							
Shortleaf pine	8.4	8.5	3.6	4.0	1.54	<0.01	0.11	0.27
Oaks	11.6	11.0	10.0	9.5	1.08	0.06	1.05	0.36
Other canopy trees	10.0	10.0	9.0	8.5	1.58	0.09	2.14	0.39
Midcanopy trees	11.8	14.2	12.6	9.9	0.74	0.02	12.45	0.90

[a] Mean square error (MSE).
[b] Probability level (*P*).

for groundline diameter and crown width but not for height. For the other canopy trees and midcanopy trees, regeneration also tended to be slightly larger under the pine-overstory treatment, but differences were not consistently significant.

Hardwood regeneration was larger than shortleaf pine regeneration after the fifth growing season. This reflected a difference in the principal reproductive strategy of the two groups—seeds for pines versus advanced regeneration and sprouts for hardwoods. Experience elsewhere has shown that the height growth of free-to-grow pine regeneration will eventually exceed that of hardwoods on most upland sites (Wahlenberg 1960), and the 5-year results of this study seem to confirm this. At 3 years, hardwoods were 2.0 times taller than pines for the pine-overstory treatment and 3.8 times for the pine-hardwood treatment, but at 5 years the differences had diminished to 1.4 times for the pine-overstory treatment and 2.6 for the pine-hardwood treatment.

Understory Coverage

Effects of the overstory treatment on understory coverage was variable among species groups, with significant differences occurring for forbs, hardwoods, and total vegetation (table 4). Differences for grasses and pines were nearly significant. Coverage was greater in the pine-overstory treatment than in the pine-hardwood treatment: 2.3 times for grasses, 1.8 for forbs, 1.5 for hardwoods, and 3.7 for pines. In contrast, coverage of vines and shrubs was only slightly affected by overstory treatment. Differences in coverage between hardwood control methods were small and only significant for herbs and total vegetation. Herbs had slightly greater coverage in the chemical control treatment. This may be the result of reduced coverage of hardwoods and shrubs, which were the only groups treated with herbicide. However, it may also be an anomaly because the coverage of shrubs and hardwoods was only slightly lower for chemical hardwood control, and these treatment differences were not significant. Total vegetative coverage for the chemical treatment averaged 70 percent compared with 65 percent for the manual treatment.

Coverage of pines and hardwoods showed large increases in the pine-overstory treatment between 3 and 5 years, while the other groups remained fairly constant. Shelton (1997) reported that hardwoods averaged 27 percent coverage at 3 years and pines 2 percent. Thus, increases from 3 to 5 years were 1.5 times for hardwoods and 3.8 times for pines. This observation suggests that pines and hardwoods will dominate subsequent patterns of successional change in the understory.

Merchantable Tree Growth

Hardwood retention significantly reduced the growth of merchantable-sized pines: by 31 percent for basal area, 30 percent for total merchantable volume, and 28 percent for sawtimber volume (table 5). Annual growth in pine sawtimber volume averaged 135 board feet (Doyle) per acre in pine-overstory treatment and 95 board feet per acre in pine-hardwood treatment. Unthinned shortleaf pine stands are expected to be growing 260 board feet per acre per year on similar sites and ages as this shelterwood stand (U.S. Department of Agriculture 1929). Thus, the shelterwood cutting reduced pine growth by about one-half for the pine-overstory treatment and about two-thirds for the pine-hardwood treatment. Hardwood growth was surprisingly high after harvesting and actually exceeded that of shortleaf pine by 45 percent for basal area and 24 percent for total merchantable volume. The different growth rate between pines and hardwoods may reflect the younger age of the hardwoods (a mean of 61 years for hardwoods versus 66 years for pines) or inherent differences in growth patterns and ability to respond to release. Although hardwoods contributed to total merchantable growth, they did not contribute to the stand's sawtimber growth because of their small size and generally low quality. Annual mortality of shortleaf pines was very low after harvesting, averaging only 0.04 trees per acre.

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Results 5 years after harvesting showed that the shelterwood method can effectively regenerate mixed pine-hardwood stands in the Ouachita Mountains when combined

Table 4—Coverage of understory vegetation in shelterwood stands with pine and pine-hardwood overstory compositions and manual and chemical methods of submerchantable-hardwood control

	Pine overstory		Pine-hardwood overstory		Overstory treatment		Hardwood control treatment	
Species or group	Manual	Chemical	Manual	Chemical	MSE[a]	*P*[b]	MSE	*P*
	- - - - - - - - *Coverage* (*percent*) - - - - - - - -							
Grasses	20.6	24.7	11.5	9.4	66.4	0.06	38.7	0.76
Forbs	0.7	2.2	0.5	1.1	0.1	0.01	0.4	0.02
Vines	15.9	13.3	12.1	12.0	29.8	0.42	4.3	0.25
Shrubs	13.4	10.3	18.6	13.8	20.6	0.15	28.6	0.19
Hardwoods	43.8	37.0	25.3	28.5	52.9	0.03	50.2	0.62
Pines	7.9	9.7	2.2	2.5	20.0	0.06	4.6	0.35
Total vegetation[c]	80.5	75.5	58.6	54.9	1.6	<0.01	13.3	0.05

[a] Mean square error (MSE).
[b] Probability level (*P*).
[c] Coverage of total vegetation is less than the sum of individual species groups because of multiple occupancy.

Table 5—Mean annual net growth of merchantable-sized trees in shelterwood stands with pine and pine-hardwood overstory compositions and manual and chemical methods of submerchantable-hardwood control

	Pine overstory		Pine-hardwood overstory		Overstory treatment		Hardwood control treatment	
Species or group	Manual	Chemical	Manual	Chemical	MSE[a]	*P*[b]	MSE	*P*
	- - - - - - - - -*Basal area* (*ft*^{2}*/acre*)- - - - - - - - -							
Shortleaf pine	0.85	0.84	0.55	0.62	0.007	<0.01	0.018	0.64
Hardwoods	0	0	1.01	1.10	—[c]	—	0.019	0.43
	- - - - -*Merchantable volume* (*ft*^{3}*/acre*)- - - - -							
Shortleaf pine	27	27	18	20	6.3	<0.01	15.1	0.73
Hardwoods	0	0	24	26	—	—	7.4	0.36
	- - - - - -*Sawtimber volume* (*ft*^{3}*/acre*)- - - - - -							
Shortleaf pine	29	29	20	22	4.0	<0.01	12.6	0.86
Hardwoods	0	0	0	0	—	—	—	—
	- - - - -*Sawtimber volume* (*fbm/acre*)- - - - -							
Shortleaf pine	137	131	94	100	158	0.01	228	0.98
Hardwoods	0	0	0	0	—	—	—	—

[a] Mean square error (MSE).
[b] Probability level (*P*).
[c] Too few hardwoods existed in sawtimber size class for analysis.

with low-cost, low-impact site preparation that controls the submerchantable trees left after harvesting. Despite substantial undisturbed litter and slash after harvesting and hardwood control, enough favorable microsites were available to establish acceptable shortleaf pine and hardwood regeneration. Pine regeneration principally came from seeds dispersed after treatment, but hardwoods developed from advanced reproduction and sprouting. This difference in reproduction strategy gives hardwoods an initial growth advantage. However, pines generally grow rapidly after establishment, providing acceptable regeneration when density and stocking are at levels similar to those reported here.

Retention of overstory hardwoods within a shelterwood pine stand will have the most significant impact on environmental conditions in the understory. Such hardwoods appear to suppress development of pine regeneration to a greater degree than an equivalent pine basal area, reflecting differences in the crown features of the two species groups. The limit for retaining hardwoods within a pine shelterwood stand appears to be fairly low. Results suggest that 15 square feet per acre of hardwood basal area can be retained in a scattered distribution for at least the first 5 years after the reproduction cut. This seems logical based on the generality that hardwoods produce about twice the overstory competition as the same pine basal area. An overstory basal area of 30 square feet per acre of pines and 15 square feet per acre of hardwoods is equivalent to a pine overstory 60 square feet per acre, which is generally considered to provide an acceptable environment for the development of pine regeneration in uneven-aged stands (Baker and others 1996). However, the long-term success of either overstory treatment is doubtful unless overstory trees are removed entirely or periodically reduced to acceptable stocking levels. Although subsequent harvesting will damage existing regeneration, Grano (1961) found that the loss in pine milacre stocking was only 10 percent for a basal area removal of 31 square feet per acre and 16 percent for 42 square feet per acre.

The upper limit for acceptable overstory stocking has not been well established for even-aged reproduction cutting methods, probably because traditional guidelines call for overstory removal as soon as regeneration reaches acceptable levels. Undoubtedly, the pine-hardwood overstory treatment of this study will reach the upper limit for acceptable stocking much sooner than the pine-overstory treatment. Based on observed growth rates, basal areas in the pine-overstory treatment should take well over 20 years to reach 75 square feet per acre, which is considered the upper stocking level for adequate development of pine regeneration in uneven-aged pine stands (Baker and others 1996). Because of its higher initial stocking, the pine-hardwood overstory treatment should reach this limit in 7 years, when projected basal areas are 35 square feet per acre for pines and 20 square feet per acre for hardwoods (equivalent to a pine basal area of $35 + 2 \times 20 = 75$ square feet per acre). Of course, this prediction needs confirmation by the continued monitoring of overstory and understory dynamics, but the general contrast between overstory treatments is clear: the pine-hardwood overstory treatment stocking must be reduced within 5 to 10 years after harvest to sustain the development of regeneration.

Because merchantable hardwoods were removed in this study during harvesting, subsequent control treatments were low in intensity and cost. The chemical control treatment was restricted to stumps of individual stems in the 1- to 5-inch d.b.h. classes and, therefore, was applied to only a fraction of the stand's hardwoods. The herbicide effectively controlled sprouting in some but not all species. Early results suggest that chemical control was not justified in the conditions tested here because it failed to substantially improve the amount or size of shortleaf pine regeneration. Other conditions had more influence on the acceptable establishment of shortleaf pine regeneration, including abundant pine seed production and low initial levels of competing vegetation.

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SMALL MAMMAL COMMUNITIES OF STREAMSIDE ZONES IN THE OUACHITA MOUNTAINS

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Natural forest stands along perennial and intermittent streams, commonly called streamside zones (SZs), frequently are retained for wildlife habitat enhancement and watershed protection when adjacent stands are harvested. However, little is known regarding wildlife habitat relationships within SZs, especially as they relate to SZ widths and influences of habitat surrounding SZs. Thus, our objective was to quantify effects of SZ width and age/structure of adjacent pine plantations on small mammal communities in the Ouachita Mountains of west central Arkansas.

We trapped small mammals in SZs of 6 width classes (0, 1-20, 21-41, 41-60, 61-100, and > 100 m) flanked by loblolly pine (*Pinus taeda*) plantations of 3 age/structural classes (young/open, closed canopy, and older/thinned) yielding a matrix of 18 stands. Three sets of these matrices were studied for 2 years each during February 1990–1995. During each 2-year sampling period we also sampled one untreated streamside forest with no adjacent plantations. For the 1990-91 sampling period we were unable to locate a > 100 m SZ within an older/thinned plantation; thus including the 3 untreated streamside forests, a total of 56 study areas were sampled on Weyerhaeuser and Ouachita National Forest lands.

Small mammals were sampled on each study area for 10 consecutive days using snap traps. Trap lines were parallel to the general stream course. Each line had 9 trap stations at 20-m intervals. Within SZs < 20 m wide, one trap line was placed on each side of and within 5 m of the stream; a second set of trap lines was placed within the plantations 40 m from each side of the stream. Small mammals were trapped only in plantation settings where the SZs were < 20 m wide. For SZs 20-60 m wide, two trap lines were again placed within 5 m of the stream; a second set of trap lines was placed within 5 m of the SZ/plantation edge (inside the SZ). For SZs > 60 m wide, we placed two trap lines within 5 m of the stream, 5 m inside the SZ/plantation edge, and about midway between the stream and SZ/plantation edge trap lines, for a total of six trap lines.

Structure of adjacent pine plantations had a noticeable effect on small mammal numbers, richness, and diversity. SZs flanked by older/thinned and young/open plantations had higher numbers, richness, and diversity than those surrounded by closed canopy plantations. Untreated streamside forests were comparable in numbers, richness and diversity to SZs flanked by young/open and older/thinned plantations. SZs flanked by closed canopy pine plantations had the lowest small mammal numbers, richness, and diversity of all treatments.

Small mammal numbers tended to increase with SZ width. However, richness and diversity varied little with SZ width. Numbers, richness, and diversity in untreated streamside forests were comparable to other SZ widths. Small mammal species composition varied with SZ width. Species like the fulvous harvest mouse (*Reithrodontomys fulvescens*) and the woodland vole (*Microtus pinetorum*) tended to be associated more with narrower SZs. *Peromyscous* spp. tended to be more associated with wider SZs.

Small mammal richness and diversity did not increase with increasing SZ width, but was influenced by the structure of surrounding pine plantations. Young/open and older/thinned plantations likely support more species of small mammals than closed canopy plantations. Species from adjacent plantations with these structures could also utilize SZ habitat, particularly along edges. Thus, within pine-plantation dominated landscapes, SZs of a variety of widths embedded in plantations of different age/structural characteristics should enhance overall small mammal diversity.

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AVIAN COMMUNITIES OF STREAMSIDE ZONES IN THE OUACHITA MOUNTAINS OF ARKANSAS

Ronald E. Thill, Philip A. Tappe, M. Anthony Melchiors, and T. Bently Wigley[1]

Linear strips of forest along intermittent and perennial streams, commonly called streamside zones (SZs), are frequently retained for watershed protection and wildlife habitat enhancement in southern pine forests when adjacent stands are harvested. However, little is known regarding wildlife communities associated with SZs, particularly in relation to varying SZ widths and interactions with surrounding habitat matrices. The objective of this study was to assess relationships between forest bird communities and SZs of various widths within pine plantations of several age/structural classes within the Ouachita Mountains of Arkansas.

We studied breeding bird communities in SZs of 6 width classes (0-246 m) flanked by loblolly pine (*Pinus taeda*) plantations of 3 age/structural classes (young/open, closed canopy, and older/thinned) yielding a matrix of 18 stands. Three sets of these matrices were studied for 2 years each between May 1989 and February 1995. During each 2-year sampling period, we also sampled one untreated riparian forest that had no adjacent plantations. We were unable to locate the widest class SZ within a thinned plantation during the 1989-90 sampling period; consequently, we sampled a total of 53 SZ/plantation settings plus 3 untreated controls. Data were generally obtained from one 80-m-wide by 200-m-long strip transect centered on each stream. Thus, for SZs averaging < 80 m in total width, varying proportions of SZ and plantation habitat were sampled. Six observers sampled each area for 30 minutes twice each spring between early May and mid-June.

Nearctic-Neotropical migrants comprised > 50 percent of birds and species encountered, regardless of SZ width. Although bird abundance, species richness, and diversity varied little with SZ width, bird composition changed with increasing SZ width. Birds frequenting narrower SZs were mainly those associated with young brushy stands and edge habitat, while those frequenting wider SZs were mostly species associated with mature pine-hardwood and bottomland hardwood forests. Avian abundance, richness, and diversity within SZs flanked by plantations were generally comparable to or higher than those found in the three untreated controls. These measures tended to be higher when SZs were flanked by young/open plantations and lower when flanked by closed-canopy plantations. Brown-headed cowbirds (*Molothrus ater*) were more abundant in narrower SZs. Streamside zone policies should accommodate species strongly associated with riparian habitat and with high conservation priority. Management policies that promote a variety of SZ widths, related to stream and floodplain characteristics, should enhance avian diversity within pine-plantation-dominated landscapes.

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INITIAL WILDLIFE HABITAT RESPONSES TO ALTERNATIVE FOREST REGENERATION METHODS IN THE OUACHITA MOUNTAINS

Ronald E. Thill, Roger W. Perry, Nancy E. Koerth, Philip A. Tappe, and David G. Peitz[1]

In conjunction with phase II bird and small mammal studies, we measured an array of habitat features in 20 stands representing 4 replications of 4 pine regeneration treatments (clearcut/plant, pine-hardwood shelterwood, pine-hardwood single-tree selection, and pine-hardwood group selection) plus 4 untreated controls. At study initiation, all stands were > 60 years old, > 34.6 acres in size, located on southerly aspects, and had slopes generally < 20 percent. Stands were harvested during the summer of 1993. Habitat measurements were collected during late winter of 1995, 1997, and 1999. Here we summarize initial responses of several key wildlife habitat parameters to these five treatments.

Most habitat measurements were obtained at 30 permanent sampling points in each stand. Using plots of various sizes, we ocularly estimated percent cover of forbs, graminoids (grasses and grasslike plants combined), and woody plants, and volume of down wood with an average diameter ≥ 4 inches. Densities of snags (standing dead trees ≥ 3.28 feet tall) were tallied within 49.2-foot-wide belt transects that traversed each area and averaged 4,833 feet in total length.

None of the treatments resulted in major increases in evergreen forbs or woody plants during late winter; however, clearcut and shelterwood stands averaged four to five times more evergreen woody plant cover than controls by the sixth year after treatment. Evergreen graminoid cover increased substantially under all regeneration treatments compared to controls. Total volume of down wood within harvested stands was consistently two or more times greater than that of control stands for all 3 sampling years. Pines and hardwoods contributed equally to down wood volumes. Snag densities decreased over time under all treatments, indicating that snag recruitment from insects, diseases, and lightning strikes is not keeping pace with natural losses. Unfortunately, 44 to 71 percent of the snags were too small [< 6 inches diameter at breast height (d.b.h.)] for cavity excavation by woodpeckers. Snags 6.0 to 13.7 inches d.b.h. may be utilized as cavity trees by smaller cavity-nesting birds, but pileated (*Dryocopus pileatus*), red-bellied (*Melanerpes carolinus*), and red-headed woodpeckers (*M. erythrocephalus*) generally require snags >13.7 inches d.b.h. When averaged across years, total densities of 6.0- to 13.7-inch snags (pine and hardwood) ranged from 1.6 per acre (shelterwood) to 4.6 per acre (group selection), while densities of snags > 13.7 ranged from just 0.3 per acre in shelterwood stands to 0.5 per acre in group selection stands. Retention of unharvested greenbelt habitat should eventually result in higher densities of larger snags.

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SHORTLEAF PINE SEED PRODUCTION FOLLOWING PARTIAL CUTTING IN THE OUACHITA MOUNTAINS

Robert F. Wittwer and Michael G. Shelton[1]

Abstract—Seed production is one of the principal determinants of successful natural regeneration of shortleaf pine (*Pinus echinata* Mill.) in both uneven-aged and even-aged silvicultural systems. In this paper, we describe the amount and periodicity of shortleaf pine seed production observed in a number of stands with monitoring periods of up to 8 yr. Results were compiled from: (1) two replicated research studies testing hardwood retention in uneven-aged and even-aged reproductive cutting methods in the eastern part of the Ouachitas, and (2) eight operational-level stands, which were mostly located in the western part of the Ouachitas. The following generalities can be drawn from the combined results: (1) seed production is highly variable, ranging from zero in some years to over 3 million sound seeds per ac in others, (2) reduced seed production occurs during some years in stands where midcanopy hardwoods are present but the effects are not consistent from year to year, (3) seed production tends to decrease in older stands and in overstocked stands, and (4) the western part of the Ouachitas tends to have more failures and fewer good seed crops. Results indicate that seed production will usually be adequate for natural regeneration of shortleaf pine within most of the Ouachita Mountains.

INTRODUCTION

Although the most widespread of the southern pines, the greatest concentration of shortleaf pine (*Pinus echinata* Mill.) is in the Ouachita Mountains of central Arkansas and eastern Oklahoma (Lawson 1990). Within this region, shortleaf pine is the most important commercial species, and it is the dominant naturally occurring pine species. Shortleaf pine can be successfully regenerated by both artificial and natural methods. One of the most critical determinants of successful natural regeneration of shortleaf pine is an adequate seed supply (Baker 1992, Lawson 1986). Some silvicultural strategies, such as retaining fruitful trees and promoting their general vigor, can be used to enhance seed production within a stand, but these techniques are overshadowed by the uncontrollable influences of seed and cone consumers and weather. Considering these restrictions, resource managers relying on natural regeneration of this important species should be familiar with the periodicity of seed crops.

The timing of an adequate seed supply with a receptive seedbed and low levels of competing vegetation is the greatest challenge to managers relying on natural reproduction cutting methods (Shelton and Cain, in press). In this paper, we compile data on shortleaf pine seed-production from two research studies and eight operational stands; all sites are located in the Ouachita Mountains. This information will provide land managers with knowledge about the periodicity of seed crops within the Ouachitas and the extent to which they can enhance seed production through silvicultural manipulations.

The Pine Reproductive Cycle

For shortleaf and the other southern pines, the total time between strobili initiation and seed dispersal is over 2 yr. The wide variation observed in annual shortleaf pine seed crops can be attributed to several factors. Flower production in an old-field stand was observed to vary by a factor of eight times over a 6-yr period in the Virginia Piedmont (Bramlett 1972). Flower initiation is influenced by the interaction of physical environmental factors (nutrients, moisture, light, temperature) and physiological processes (Barnett and Haugen 1995).

Subsequent mortality is caused by spring frosts, insects, and physiological abortion. Spring frosts, late enough to damage new juvenile leaves on hardwoods, have been observed to damage female flowers (Campbell 1955, Hutchinson and Bramlett 1964). Male flowers seem less susceptible to frost damage. Overall survival of flowers to mature cones varied from 3 to 65 percent and averaged 29 percent over a 6-yr observation period (Bramlett 1972). Greatest losses occurred between May and September during the first year of cone development. Mortality during the second year was attributed to squirrels and insects. Yearian and Warren (1964) found 39 insect species to be associated with shortleaf and loblolly pine (*Pinus taeda* L.) conelets and cones in Arkansas; *Dioryctria clarioralis* (Walker) and *D. amatella* (Hulst) were the most destructive. Seed sampling in 22 stands included in the Phase I, Ecosystem Management Research pilot study found about 12 percent of the seed to be damaged by seed bugs (Mangini and others 1994); *Leptoglossus corculus* (Say) and *Tetyra bipunctata* (Herrich-Schaffer) caused the most damage.

Shortleaf pine seed yields have been found to vary from failures to over two million per acre in previous studies (Wittwer and Shelton 1992). Three long-term studies have been previously conducted: (1) a 10-yr study in the Piedmont region of the Carolinas and Georgia in the eastern portion of the natural range (Bramlett 1965), (2) a 10-yr study in east Texas in the southwest portion of the range (Stephenson 1963) and (3) a 9-yr study in the Ozark and Ouachita Mountains of Arkansas, Missouri, and Oklahoma in the northwestern portion of the natural range (Shelton and Wittwer 1996). In general, all studies recorded three or four

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good seed crops during the study periods. In the southeastern Piedmont, seed yields were observed to decrease with increasing latitude (Bramlett 1965). In the Ozark/Ouachita Mountain study, yields were lower in the western portion of the region, an area representing the northwestern limit of shortleaf's natural range (Shelton and Wittwer 1996). Good seed crops may not be distributed at regular intervals; two successive years with good yields may be followed by three or four poor years.

Seed quality, expressed as the percentage of the total crop that is sound, appears to increase with the higher yields. Bramlett's (1965) 10-yr study in the Piedmont found 57 percent sound seed in the three good seed crops and 41 percent for the other seven crops. In east Texas, Stephenson (1963) found sound seed to average 61 percent for the four years with good crops during the 10-yr study. Shelton and Wittwer (1996) found the percentage to range from about 30 percent in poor seed years to 70 percent for bumper seed crops. Intensive cone sampling from two Ouachita Mountain stands in a year with a mediocre seed crop found between 30 and 50 percent of the seeds to be sound (Wittwer and others 1997).

METHODS

Study Areas and Treatments

Study 1 (uneven-aged pine-hardwoods)—This study was installed in the Winona Ranger District of the Ouachita National Forest in Perry County, AR (Shelton and Murphy 1997). Plots were oriented along an east-west ridge, which is typical of the physiography of the Ouachita Mountains. Elevations ranged from 640 to 790 ft above sea level. Slopes of individual plots ranged from 8 to 21 percent. Soils of the study area are mapped as the Carnasaw and Pirum series, both Typic Hapludults. These are well-drained, moderately deep soils that developed in colluvium and residuum weathered from sandstone and shale. Natural fertility and organic matter are low, and the soils are strongly acidic. Site index for shortleaf pine averaged 57 ft at 50 yr and ranged from 53 to 64 ft, which is typical of upland sites in the Ouachita Mountains (Graney 1992). Annual precipitation averages 52 in., and temperature averages 40^0 F in the winter and 79^0 F in the summer (U.S. Department of Commerce 1968). Winter is the wettest season and autumn is the driest. Water deficits typically develop during the summer.

Vegetation in the study area was typical of much of the forested landscape in the Ouachita Mountains, where upland forests are dominated by shortleaf pine and mixed oaks (Guldin and others 1994). Overstory basal area (trees > 3.6 in. d.b.h.) averaged 90 ft^2 per ac for shortleaf pine and 32 ft^2 per ac for hardwoods before study implementation. Oaks accounted for 84 percent of the total hardwood basal area. Overstory pines and oaks in the initial stand ranged in age from 30 to over 110 yr (Shelton and Murphy 1991); dominant pines averaged 76 yr old, while dominant oaks averaged 70 yr old. No silvicultural treatments had been applied within a decade of study installation.

Sixteen square 0.50-ac plots were established and surrounded by a 58-ft isolation strip that was treated in an identical manner. Plots were arranged in a randomized complete block design with four replicates. One plot per block was designated as a control that was not treated in any manner. The pine component for the treated plots was reduced to a target basal area of 60 ft^2 per ac using uneven-aged marking guidelines for single-tree selection (Baker and others 1996). An average of 81 percent of residual pines were > 10 in. d.b.h., which is usually accepted as the minimum size for high-seed producing potential (Shelton and Wittwer 1995). Plots were harvested during the winter of 1988-89. Imposed treatments were three targeted levels of retained overstory hardwoods (0, 15, and 30 ft^2 per ac in trees > 4 in. d.b.h.). Hardwoods (> 0.6 in. d.b.h.) that were not designated for retention were controlled with stem-injected herbicide.

Within each plot, four 0.9-ft^2 seed traps (Cain and Shelton 1993) were located about 30 ft from the center of the plot in a square pattern and about 100 ft from the outer boundary of the plot. Seed collections were generally made during the middle and end of each October-to-February period, which is the normally accepted period for dispersal of shortleaf pine seed (Wittwer and Shelton 1992). Monitoring began in October 1988 for the treated plots and in October 1990 for the untreated control, and the last collection was made in March 1997.

Study 2 (shelterwood pine-hardwoods)—This study area was located about 0.5 mile from study 1, and both were very similar in stand and site conditions (Shelton 1997). Elevations ranged from 640 to 810 ft above sea level. Slopes of individual subplots ranged from 5 to 26 percent. Soils were mapped as the Carnasaw and Pirum series. Site index for shortleaf pine averaged 60 ft at 50 yr, ranging from 56 to 65 ft, and the dominant pines averaged 66 yr old. No silvicultural treatments had been applied within a decade of study installation. Pre-treatment overstory basal area averaged 74 ft^2 per acre for shortleaf pine and 41 ft^2 per acre for hardwoods. Oaks accounted for 92 percent of the hardwood basal area.

Eight rectangular 1.7-ac plots were established and surrounded by a 60-ft isolation strip that was treated in an identical manner. Plots were arranged in a randomized complete block design with four replicates. The pine component on all plots was reduced to a basal area of 30 ft^2 per ac in trees selected principally for their potential as seed trees; all trees were > 10 in. d.b.h. Retention of overstory hardwoods was 0 and 15 ft^2 per ac in trees > 4 in. d.b.h. All merchantable pines were harvested during the winter of 1989-90, and merchantable hardwoods not designated for retention were harvested during the spring and summer of 1990. After harvesting was completed, plots were split into halves, and two control methods for submerchantable hardwoods (0.6 to 3.5 in. d.b.h.) were imposed (chain-saw felling with and without a stump-applied herbicide). Three 0.9-ft^2 seed traps were located along the center line of each split plot; traps were at least 120 ft from the untreated stand and 100 ft from adjacent whole plots. Monitoring began in October 1990, and the last collection was made in March 1997.

Operational stands—A 3-ac area within an operational seed-tree stand was selected for monitoring in the eastern Ouachitas. The area was located about 2 miles from studies

1 and 2 and was similar to those studies in soil and site conditions. Elevations ranged from 520 to 600 ft. The seed-tree cut was made in 1985, and hardwoods were controlled by chain-saw felling. Nine 0.09-ft^2 seed traps were systematically located within the area. Monitoring began in October 1993 and the last collection was made in March 1997.

Five natural stands on the Ouachita National Forest (ONF) and two stands on Choctaw National Forestry (CNF) lands in eastern Oklahoma were selected for monitoring (October 1989 through February 1997). Stands were 22 to 40 ac in area. The ONF stands were 60-80 yr old when sampling began and had been subjected to even- or uneven-aged reproduction cutting methods. The CNF stands were 45-50 yr old when sampling began; one was thinned while the other was not. Stands on the ONF were in LeFlore County, OK, while those on CNF lands were in Latimer County, OK. Elevations ranged from 800 to 1200 ft above sea level. The Carnasaw and Bengal soil series were common in the stands, although other series were present. All soils were moderately deep or deep and moderately well-drained or well-drained. Texture of the surface horizon ranged from stony- to fine-sandy loams. Annual precipitation averages approximately 45 in., and temperature averages 43^0 F in the winter and 80^0 F in the summer (U.S. Department of Commerce 1968). Seed production was sampled with six to ten, 2 by 2-ft wood-frame and wire mesh traps in each stand (Scholtens 1979).

Seed Processing

Coarse litter was generally removed from the seed traps in the field, and fine litter and seeds were brought into the laboratory for separation. Collected material was refrigerated until processed. Seeds were counted, and viability was generally determined by cutting seeds and inspecting the contents (Bonner 1974). Seeds with full, firm, undamaged, and healthy tissue were judged to be potentially viable and were tallied as sound seeds. Although ineffective for stored seeds, the cut test can be accurate when applied to fresh seeds (Bonner and others 1994). To confirm validity of the cut test in our studies, a subsample from the 1993 and 1994 collections for studies 1 and 2 totaling 1,800 seeds was germinated under controlled conditions following 30 days of cold/moist stratification. At the end of the 30-day germination periods, a cut test revealed that only 3.5 percent of the ungerminated seeds were full, indicating that virtually all of the full seeds collected in our studies were viable. Because of the large number of seeds to be processed from the Oklahoma stands in 1993, seeds were considered sound if they sank in ethanol (Krugman and Jenkinson 1974). A subsample of these seeds was cut and yielded the same results as the float test.

Data Analysis

Means were calculated for each plot or subplot for study 1 and 2 and for the entire stand for the operational stands. For study 1, data were analyzed using analysis of variance for a randomized, complete block design. Differences among treatment means were isolated by using the Ryan-Einot-Gabriel-Welsch Multiple Range Test at a probability level (P) of 0.05. This procedure, which is one of the most powerful step-down, multiple-range tests available, controls the experiment-wise error rate (SAS Institute 1989). For study 2, data were analyzed by analysis of variance for a 2 by 2 factorial, split-plot randomized complete block design using the SAS procedure GLM (SAS Institute 1989). Since there were only two levels for each factor, means were not separated but were presented with the associated mean square error (MSE) and P. Significance was accepted at $P < 0.05$.

RESULTS AND DISCUSSION

Seed Yield

Seed yields exhibited wide annual variation, ranging from complete failures in some stands for some years to over 3 million sound seeds per ac in the pine-only shelterwood stands in 1993 (tables 1, 2, and 3). In the eastern Ouachitas, five of the eight seed crops sampled in both shelterwood and uneven-aged stands produced generally good yields (tables 1 and 2). An adequate seed crop to successfully regenerate shortleaf pine has usually been specified as being in excess of 80 thousand seeds per ac (Baker 1992, Haney 1962, Shelton and Wittwer 1996). Suitable conditions for germination and seedling establishment may be present for three years after site preparation, perhaps longer on poor sites and less on more productive sites (Shelton and Wittwer 1992). In the eastern Ouachita Mountain stands, an adequate crop was produced at least once within any three year interval.

Hardwood retention generally had a negative influence on shortleaf pine seed production, but the effects varied from year to year. In the uneven-aged stand where hardwoods were retained, seed production was significantly decreased during 1993 and 1994 and for the 6-year mean (table 1). A similar significant decrease occurred in the shelterwood stand during three seed crops (1993, 1994, and 1996) and for the 7-yr mean (table 2). The negative effects of hardwoods on seed production were usually greatest for the better seed crops. The method of controlling submerchantable hardwoods did not significantly affect seed production in the shelterwood stand.

The importance of reducing shortleaf pine basal area was apparent in the uneven-aged stand in 1992 and 1996 (table 1). The harvested plots with high hardwood retention produced three times more seeds than the unharvested controls in 1992 and five times more in 1994. These two areas had about the same hardwood basal area, but the pine basal area was about 30 ft^2 per ac lower in the harvested plots. Overall, the harvested plots with no hardwoods produced about twice the sound seeds as the unharvested controls over the 6-yr period when both areas were monitored.

There was wide stand-to-stand variation for the operational stands sampled in the western Ouachita region for the 8-yr period from 1989 through 1996 (table 3). Only 1993 can be characterized as producing a good seed crop, but even then, three of the seven stands in the western Ouachita region exhibited failures. This trend agrees with a 9-yr study of seed production throughout the Interior Highlands of Arkansas, Missouri, and Oklahoma from 1965 through 1973 (Shelton and Wittwer 1996). Seed crops were only adequate in the western Ouachitas during 1966 and 1967, when bumper seed crops occurred throughout the rest of the region.

Table 1—Annual shortleaf pine seed production after the initial harvest implementing uneven aged silviculture in a shortleaf pine hardwood stand in the Ouachita Mountains (study 1)

	Hardwood retention[a]					
Seed	None	Medium	High	Untreated control	Mean square error	P > F
year	*- - - - thousands of sound seeds per acre*[b] *- - - -*					
1989	197	144	200	—	6.78E3	0.58
1990	3	3	3	—	3.48E1	1.00
1991	75	75	68	31	8.05E2	0.16
1992	166	260	122	44	2.44E4	0.32
1993	2,654a	1,912ab	1,590ab	964b	3.73E5	0.02
1994	310a	75b	128b	20b	4.36E3	< 0.01
1995	0	6	0	0	1.31E1	0.09
1996	2,313	1,915	1,796	1,596	3.59E5	0.43
Mean[c]	920a	707ab	617ab	442b	3.37E4	0.03

[a] After harvest, merchantable basal areas (trees >3.6 inches d.b.h.) in 1988 averaged 62 square feet per acre for shortleaf pine and 0, 16 and 31 square feet per acre for none, medium, and high hardwood retention treatments, respectively. Unharvested control plots averaged 94 and 32 square feet per acre for pine and hardwood basal areas, respectively, in 1990, and there were also 8 square feet per acre of understory hardwoods (0.6 to 3.5 inches d.b.h.).

[b] Row means followed by different letters are significantly different.

[c] Means are from 1991 to 1996 so that control plots could be compared to treated plots.

Table 2—Annual shortleaf pine seed production in shelterwood stands with two overstory compositions and two control treatments for submerchantable hardwoods (study 2)

	Pine only[a] overstory		Pine hardwood[a] overstory		Overstory composition		Submerchantable hardwood treatment	
Seed	Manual	Chemical	Manual	Chemical	MSE	P > F	MSE	P > F
year	*- - - thousands of sound seeds per acre - - -*							
1990	67	50	21	17	2.71E3	0.23	2.50E2	0.23
1991	33	25	17	37	5.28E2	0.87	3.66E2	0.54
1992	142	171	121	175	3.48E3	0.80	3.70E3	0.22
1993	3,442	3,484	2,295	2,291	1.27E5	0.01	1.27E5	0.94
1994	1,431	1,596	517	572	3.14E5	<0.01	5.35E5	0.38
1995	8	17	0	4	6.38E1	0.08	2.26E2	0.44
1996	1,888	2,378	1,652	1,694	3.34E4	0.02	9.32E4	0.13
Mean	1,002	1,103	660	684	4.59E3	<0.01	1.57E4	0.36

[a] After harvesting in 1990, shortleaf pine merchantable basal area (trees >3.6 inches d.b.h.) averaged 28 square feet per acre, and hardwoods averaged 16 square feet per acre when retained.

Seed Quality

The percentage of sound seed was positively related to total seed production in the sampled stands (fig. 1). Values ranged from < 20 percent in years with the lower seed production (< 100,000 seeds per ac) to over 60 percent in good seed years. A 9-yr study conducted between 1965 and 1973 found sound seed to be about 30 percent of total production in poor seed years and 70 percent of total production in years with good crops (Shelton and Wittwer 1996). Void and defective seeds result from several factors, which are not greatly affected by management in natural stands. These factors include lack of pollen, lethal gene combinations, self-pollination, insect damage, and climatic factors (Fatzinger and others 1980).

CONCLUSIONS

Shortleaf pine seed production is highly variable for natural stands in the Ouachita Mountains. Annual production of sound seed during the 8-yr observation period from 1989 through 1996 ranged from zero to over 3 million seeds per ac. The influence of retained hardwoods varied from year to year. For the better seed crops, stands with retained

Table 3—Annual production of shortleaf pine seed from 1989 through 1996 for operational stands representing a range of residual shortleaf pine basal areas in the Ouachita Mountains

Stand	Shortleaf pine basal area	Sound seed production by year							
		1989	1990	1991	1992	1993	1994	1995	1996
	ft^2 *per acre*	- - - - - - - - - - - - *thousands of sound seeds per acre* - - - - - - - - - - - -							
ST[a]	8	NS	NS	NS	NS	188	22	0	1,058
ST	17	0	0	0	20	4	0	0	8
SW	48	0	6	0	12	1,183	0	1	20
UEA	43	NS	39	33	NS	361	0	0	33
UEA	45	NS	2	0	4	96	0	2	15
UEA	51	NS	0	0	2	87	0	2	5
THN	62	265	0	1	NS	0	1	0	0
UTHN	98	272	0	2	NS	0	0	2	2

ST = Seed tree regeneration harvest; NS = not sampled; SW = shelterwood regeneration harvest; UEA = even-aged stand harvested to implement uneven-aged silviculture; ST, SW, and UEA stands were 70 to 80 years old; THN = a 50-year old stand thinned to the specified basal area; UTHN = an unthinned 50-year old stand.

[a] This stand is in the eastern part of the Ouachita Mountains; all others are in the western part.

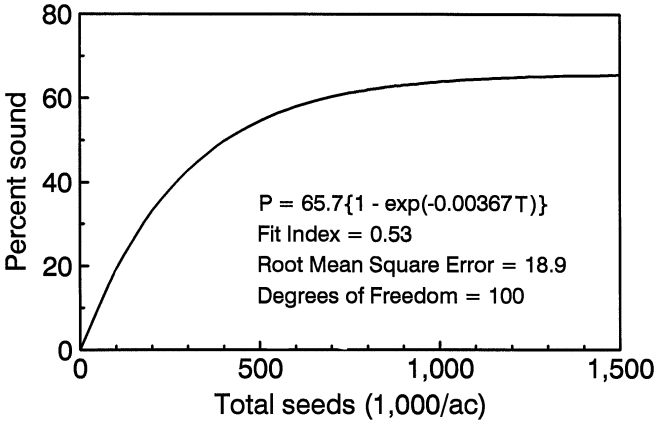

Figure 1—Relationship between sound seed percentage (P) and total seed production (T) for shortleaf pine in the Ouachita Mountains.

hardwoods tended to produce less seeds, but the difference was less evident in years with poor and mediocre seed crops. Seed production for a shelterwood stand equaled or exceeded that of a nearby uneven-aged stand, even though there was a higher basal area in trees of seed-producing sizes in the uneven-aged stand. Results of this study and previous work suggest that shortleaf pine stands in the eastern part of the Ouachita Mountains should produce adequate seed crops for successful natural regeneration using both even-aged and uneven-aged methods.

Results from this study support earlier reports of the sparse and sporadic seed crops in the western Ouachita Mountains and suggest that regional climatic factors near the western limit of shortleaf's natural range are the overriding influence. Where seed production limits the success of natural regeneration, managers should pay careful attention to: (1) retention of an adequate density of trees with a high potential for seed production, (2) individual tree characteristics related to seed production, such as crown class, vigor, size (d.b.h. > 10 in.), and presence of old cones, (3) preparatory cutting to stimulate tree vigor, and (4) limiting the amount of retained hardwoods. Perhaps the best option is to monitor shortleaf pine's reproductive structures over the 1.5 yr period from pollination to the beginning of seed dispersal. Particular attention should be paid to the amount of maturing cones during the summer before seed dispersal (Shelton and Wittwer 1995). Monitoring can forecast when adequate seed crops are most likely to occur, and site preparation treatments can be timed to coincide.

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Cover photo: Pedestal Rock Trail View, Pedestal Rock Scenic Area,
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This volume presents 5-year results of silvicultural treatments associated with ecosystem management research in the Ouachita Mountains of Arkansas. Results from stand-level treatments include regeneration dynamics of pine and hardwood species, effects of treatment on birds and small mammals, mast production, visual quality, oak decline, and organic matter. Pretreatment landscape findings include measurements of woody vegetation; birds, mammals, and herpetofauna; fish communities and trophic structure; hydrology; and evaluation of susceptibility to gypsy moth outbreaks.

Keywords: Diversity, ecosystem management, landscape analysis, natural regeneration, pine-hardwood, reproduction cutting methods, shortleaf pine, stand-level effects, visual quality.