United States Department of Agriculture

National Best Management Practices Monitoring Summary Report

Program Phase-In Period
Fiscal Years 2013–2014

Forest Service FS-1070 December 2015

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Cover photo: *Uinta-Wasatch-Cache National Forest in Utah.*

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Contents

Tables

Figures

Member of an interdisciplinary team helps conduct Best Management Practices monitoring on the Mammoth Creek reroute project, Inyo National Forest, California.

Executive Summary

The Forest Service, an agency of the U.S. Department of Agriculture (USDA), uses Best Management Practices (BMPs) to increase accountability and to protect and maintain water resources on National Forest System (NFS) lands. BMPs are to be applied using an adaptive management strategy of implementation, monitoring, and adjustment of practices based on monitoring results (USDA Forest Service 2012).

The National BMP Program provides National Core BMPs, standardized monitoring protocols to evaluate implementation and effectiveness of the National Core BMPs, and a data management system to store and analyze the resulting monitoring data. BMP evaluations are completed by interdisciplinary teams of resource specialists and include assessments of whether BMP prescriptions were planned, implemented, and effective at meeting water resource objectives. BMP implementation and BMP effectiveness are rated separately according to a standardized rating system. A composite BMP performance rating based on the implementation and effectiveness ratings is given to evaluations in which both implementation and effectiveness assessments have been completed at the same site. Assigning a rating outcome to each BMP evaluation enables tracking of patterns and trends in BMP performance over time at multiple scales within the agency. In addition, during the field evaluations, information is gathered on site-specific actions or changes in procedures that would improve BMP implementation or effectiveness. This information can be used to adjust management practices to better protect water resources on NFS lands.

BMP monitoring has been conducted on NFS lands for many years, but there has been little consistency across regions or administrative units in how BMPs were monitored or how the data were summarized. The National BMP Program has addressed these shortcomings by providing a nationally consistent, systematic, and objective approach to BMP monitoring.

Fiscal year (FY) 2014 was the second year of a 2-year phase-in period of the National BMP Program. The purpose of the 2-year phase-in period was to familiarize Forest Service administrative units with the National BMP Program tools and procedures and to test and refine the National BMP monitoring protocols and associated rating rulesets. This report identifies the successes and results of the second year of BMP monitoring and demonstrates the capability of a consistent nationwide monitoring program to document BMP performance. With completion of the phase-in period, the National BMP Program is now in full implementation.

In FY 2014, 97 Forest Service administrative units completed a total of 600 BMP evaluations. The percentage of administrative units that completed at least one BMP evaluation increased from 74 percent in FY 2013 to 87 percent in FY 2014. While most of the completed BMP evaluations used monitoring protocols in the Road Management Activities, Recreation Management Activities, and Mechanical Vegetation Management Activities resource categories, each of the 10 resource categories had at least 28 completed BMP evaluations.

Of the 600 total evaluations, 94 percent (566) included implementation assessments, 90 percent (539) included effectiveness assessments, and 85 percent (509) included both implementation and effectiveness assessments. In all, 61 percent of the BMP implementation evaluations were rated as “Fully Implemented” or “Mostly Implemented,” 65 percent of the BMP effectiveness evaluations were rated as “Effective” or “Mostly Effective,” and 56 percent of the sites where BMP implementation and effectiveness were both monitored had composite ratings of “Excellent” or “Good.” While these data show room for improvement in BMP implementation and effectiveness across the agency, prior to development of the National BMP Program, it was impossible to report on BMP implementation and effectiveness on a national scale in a coherent, understandable, and useful way.

The best overall performance of BMP implementation and BMP effectiveness, as indicated by the percentage of evaluations rated as “Excellent” or “Good,” was in the Mechanical Vegetation Management Activities, Chemical Use Management Activities, and Wildland Fire Management Activities resource categories. Mechanical Vegetation Management Activities and Chemical Use Management Activities also had the lowest percentages of evaluations in which corrective actions or adaptive management actions to improve BMP implementation or effectiveness were identified. The resource categories with the poorest overall BMP performance were Rangeland Management Activities, Water Uses Management Activities, and Minerals Management Activities. Recreation Management Activities and Road Management Activities had the highest percentages of evaluations rated as “No Plan,” meaning no BMPs were prescribed. These latter five resource categories had high percentages of evaluations in which corrective actions or adaptive management actions were identified.

As the agency moves from the phase-in period into full implementation of the program in FY 2015, finalization of the protocols and rating system will allow trends in BMP implementation and effectiveness to be determined at local, regional, and national scales. As regions and administrative units analyze BMP monitoring results, improvement in BMP implementation and effectiveness is anticipated through (1) improved consistency in field monitoring, (2) the identification of BMP deficiencies and recommendations for corrective and adaptive management actions, and (3) improved BMP planning during project development and operation and maintenance of sites.

Rock apron below a culvert outfall to dissipate energy and disperse concentrated flow, Medicine Bow-Routt National Forests, Colorado.

Introduction

This report reviews the monitoring results of the Forest Service National Best Management Practice (BMP) Program conducted across National Forest System (NFS) lands in fiscal year (FY) 2014. FY 2014 marked the second year of the 2-year phase-in process for the National BMP Program. The purpose of the 2-year phase-in period was to familiarize Forest Service administrative units with the National BMP Program tools and procedures and to test and refine the National BMP monitoring protocols and associated rating rulesets. This report will identify the successes of this second year of BMP monitoring and demonstrate the capability of a consistent nationwide monitoring program to document BMP performance. With completion of the phase-in period, the National BMP Program is now in full implementation.

With the introduction of the National BMP Program in 2012, the Forest Service reinforced its commitment to protecting and maintaining water quality and aquatic resources on NFS lands. The Forest Service manages 193 million acres of national forests and grasslands, containing approximately 400,000 miles of streams, 3 million acres of lakes, and numerous aquifer systems that provide drinking water for approximately 124 million people (USDA Forest Service 2010). These waters also provide recreational opportunities and habitat for aquatic and riparian wildlife. Water is a vital resource to the productivity and enjoyment of our national forests and grasslands. Maintaining water quality is a critical component of the Forest Service mission to sustain the health, diversity, and productivity of the Nation's forests and grasslands to meet the needs of present and future generations. The National BMP Program is a critical component of all land-disturbing activities that have potential to affect water quality and aquatic health.

The National BMP Program allows the Forest Service to protect the chemical, physical, and biological integrity of all water bodies on NFS lands. The National BMP Program was developed to improve management of water quality on NFS lands in a manner consistent with the Federal Clean Water Act (CWA) and State and tribal water quality programs. Current Forest Service policy directs compliance with required CWA permits and State regulations. It also requires the use of BMPs to control nonpoint source pollution to meet applicable water quality standards and other CWA requirements (USDA Forest Service 2012). The National Core BMP Technical Guide, Volume 1 (USDA Forest Service 2012) is the defining document used to incorporate the National Core BMPs into planning efforts and evaluations of all proposed land and resource management activities. BMPs are specific practices or actions used to reduce or control adverse effects to water bodies from nonpoint sources of pollution, most commonly by reducing the loading of pollutants from such sources into stormwater and waterways. BMPs can be applied before, during, and after pollution-producing activities to reduce or eliminate the introduction of pollutants to receiving waters. The National Core BMP Technical Guide, Volume 2 (USDA Forest Service in prep.) provides standardized protocols for monitoring BMP implementation and effectiveness across all NFS lands. Monitoring and tracking BMPs using a consistent method improves the agency's accountability and ability to use adaptive management principles to improve BMP performance.

Interdisciplinary review team discusses possible corrective actions during Best Management Practices monitoring on a poorly drained road segment, Allegheny National Forest, Pennsylvania.

Background

The Forest Service developed the National BMP Program to improve efficiency and accountability in management of water quality and aquatic resources on NFS lands. BMPs are used to control nonpoint source pollution consistent with the requirements of the CWA, the U.S. Environmental Protection Agency, and State, tribal, and local water quality programs. Under the CWA, States and tribes are required to develop a process to identify categories of nonpoint sources of pollution and establish procedures and methods to control such sources. Every State has a Nonpoint Source Management Program and Plan that describes how to use BMPs to control levels of nonpoint source pollution. BMPs are often the primary tool for State water quality management, although their implementation may be voluntary or required, depending on State law. All national forests and grasslands have adopted BMP prescriptions consistent with or approved by State Nonpoint Source Management Programs (USDA Forest Service 2012). In States where use of BMPs is voluntary, Forest Service policy makes their use a requirement on NFS lands as outlined in Forest Service Manual (FSM) 2532 (USDA Forest Service 1990).

Development of the Forest Service National BMP Program began in 2004 and involved numerous Forest Service resource personnel at all levels of the agency and across deputy areas, including NFS, State and Private Forestry, and Research and Development. A new Forest Service land management planning rule in 2012 (36 CFR 219.8(a)(4)) required the Forest Service Chief to establish a national BMP program. In an April 2012 letter, the Deputy Chief for NFS initiated the implementation of the National BMP Program. The Forest Service strategy for controlling nonpoint source pollution on NFS lands involves identifying necessary BMPs, applying locally appropriate BMP prescriptions, monitoring and assessing their implementation and effectiveness, and utilizing results to improve future management activities and adaptive management strategies. By establishing a consistent, objective, and adaptive process for monitoring BMPs, the Forest Service aims to protect water quality at national, regional, forest, grassland, and watershed scales. Moreover, consistency will allow data to be aggregated and analyzed at any of these levels within any reporting cycle and over the long term.

The National BMP Program consists of four components: (1) a set of National Core BMPs, (2) a guide for monitoring BMP implementation and effectiveness, (3) a data management system, and (4) corresponding national direction. The National Core BMPs are grouped into 11 resource categories, including General Planning Activities (Table 1). The National Core BMPs are purposely general and nonprescriptive so that BMP prescriptions can be tailored to meet site-specific needs for water quality protection consistent with State, tribal, and local requirements. The National Core BMPs are not intended to replace preexisting State and tribal BMPs, but rather to support States and tribes by enhancing compliance with CWA requirements on NFS lands (USDA Forest Service 2012).

Table 1. National Core BMP resource categories and the corresponding number of monitoring protocols.

BMP resource category	Number of National Core BMPs[a]	Number of monitoring protocols[b]
General Planning Activities[c]	3	0
Aquatic Ecosystems Management Activities	4	2
Chemical Use Management Activities	6	3
Facilities and Nonrecreation Special Uses Management Activities	10	4
Wildland Fire Management Activities	4	2
Minerals Management Activities	8	4
Rangeland Management Activities	3	1
Recreation Management Activities	12	9
Road Management Activities	11	9
Mechanical Vegetation Management Activities	8	3
Water Uses Management Activities	6	5

BMP = Best Management Practice.

[a] National Core BMPs are described in USDA Forest Service publication FS-990a (2012).

[b] Monitoring protocols are described in USDA Forest Service publication FS-990b (in prep.).

[c] Planning is evaluated in all of the monitoring protocols.

Interdisciplinary review teams (IDTs) conduct onsite BMP evaluations to assess BMP implementation and effectiveness. Implementation evaluations provide information on the extent to which water quality protection was considered in planning and project implementation or site operation and maintenance. BMP effectiveness monitoring evaluates the extent to which BMPs met water resource management objectives.

The National BMP Program does not include direct monitoring of beneficial or designated uses of waterbodies. Scoring a BMP activity as "Not Effective" indicates the potential for adverse effects to water quality and do not necessarily indicate impairment of beneficial or designated uses by an activity.

In addition to the implementation and effectiveness questions, field evaluators qualitatively estimate the spatial extent and level of risk to water quality by recording whether potential pollutants are found outside of Aquatic Management Zones (AMZs), found inside AMZs, or delivered directly to waterbodies.

Interdisciplinary review team prepares to complete Best Management Practices monitoring for a commercial timber sale, Green Mountain National Forest, Vermont.

Objectives

The primary objectives of this report are to provide the results of FY 2014 National BMP monitoring as well as an overview of the entire FY 2013–2014 phase-in period. The purpose of the 2-year phase-in period was twofold: (1) to familiarize Forest Service administrative units with the National BMP Program tools and procedures, and (2) to test and refine the National BMP monitoring protocols and associated rating rulesets. This report will identify the successes of this second year of BMP monitoring and demonstrate the capability of a consistent nationwide monitoring program to document BMP performance.

The heart of the National BMP Program is the project or site evaluations used to monitor and assess BMP implementation and effectiveness. Implementation evaluations assess the extent to which site-specific water resource protection measures were planned and implemented on projects or sites. Implementation monitoring is focused primarily on answering the question, "Were site-specific BMP prescriptions developed during project or activity planning implemented as designed or planned?" Effectiveness evaluations determine the extent to which BMPs achieved their water resource protection objectives. In general, effectiveness monitoring is focused on answering the question, "Were the site-specific BMP prescriptions, as implemented, effective at protecting water quality and aquatic health?" To provide a consistent BMP monitoring approach across the agency, 42 BMP monitoring protocols covering the most common management projects and activities occurring on NFS lands were developed (appendix A). Each protocol evaluates one or more of the National Core BMPs. A rating ruleset unique to each protocol is used to assign a rating outcome for BMP implementation, BMP effectiveness, and a composite rating for each evaluation.

During the 2-year phase-in period, the number of completed BMP evaluations required of each administrative unit was increased from two in FY 2013 to seven in FY 2014. The number of evaluations in each resource category to be completed each year was assigned to each administrative unit by the regional offices. The allocation was based on common or characteristic management activities on each national forest and grassland with the goal of obtaining a representative distribution of evaluations in each resource category across the region. Sites to evaluate are selected either randomly from projects or activities that meet protocol-specific criteria, or nonrandomly from priority projects or activities that meet the needs of the local administrative unit. Once the National BMP Program becomes fully functioning, monitoring data from randomly selected sample sites will be used for statistical analysis of BMP evaluations at the national and regional scales.

In FY 2013, administrative units were asked to provide feedback on the protocols and rating outcomes to the National BMP Program development team so that the protocols and rulesets could be refined based on field experience. Using this feedback, in FY 2014, a small team revised the protocol questions and instructions to include clarifying language to improve execution of the protocols. The draft rulesets were changed to better reflect professional observations of BMP implementation and effectiveness at the evaluation sites.

Review of the FY 2013 BMP monitoring identified a need for more training of field resource specialists on the BMP monitoring protocols and data entry (USDA Forest Service 2015). In FY 2014, the Washington Office and the Northern Research Station partnered with the regions and forests to lead field-level National BMP monitoring training sessions. The objectives of these National BMP "train-the-trainer" sessions were to increase the understanding of the National BMP Program, continue to facilitate the use of the National Core BMPs during project planning and implementation, and to develop an interdisciplinary cadre of BMP trainers across the Forest Service regions. These training sessions were held at 12 national forests across the country between August and October 2014. Approximately 150 agency employees from 8 regional offices, 75 national forests, 3 national grasslands, and State and Private Forestry participated. Resource areas represented included hydrology, soil science, watershed management, engineering, recreation, timber management, silviculture, rangeland management, wildfire management, fish and wildlife biology, geology, minerals, and planning. In addition, over the course of FY 2014, the Washington Office and Northern Research Station provided 10 webinar-based training sessions on the data management system, including data entry, training about 170 employees.

Postfire stream crossing armoring, Inyo National Forest, California.

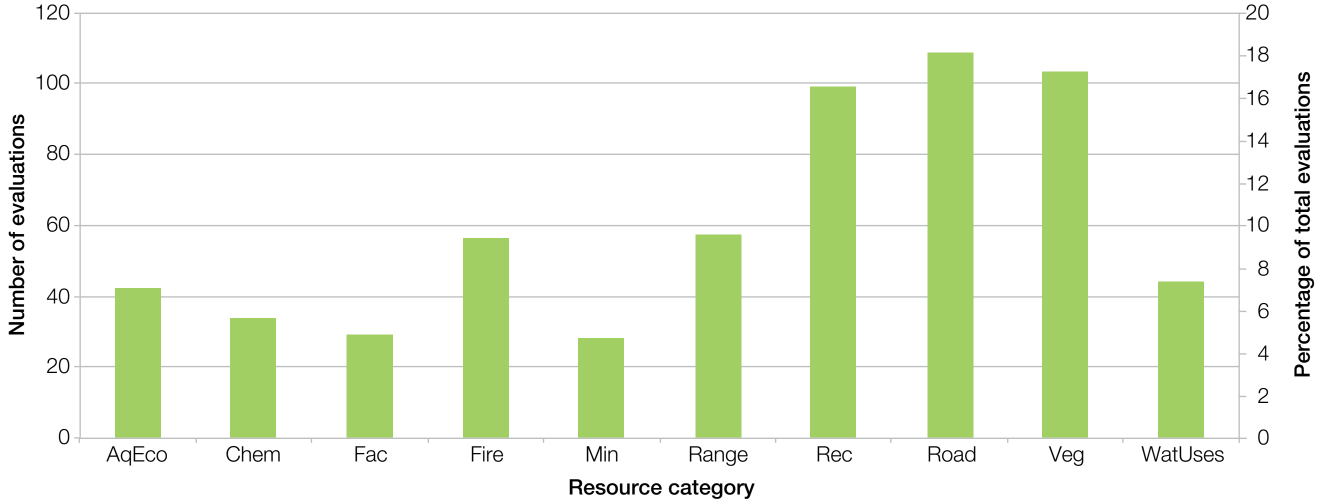
Results

BMP Evaluations Completed

A total of 600 BMP monitoring evaluations were completed during FY 2014. At least 1 BMP evaluation was completed on 87 percent (97 out of 111) of the Forest Service administrative units. The number of BMP evaluations completed for each of the 10 resource categories is shown in Figure 1. At least 28 BMP evaluations were completed for each resource category in FY 2014. Figure 1 also shows the percentages of BMP evaluations by resource category. The Road Management Activities, Recreation Management Activities, and Mechanical Vegetation Management Activities resource categories together represent more than one-half of the BMP evaluations completed in FY 2014.

Table 2 shows the number of BMP evaluations completed in FY 2014 for each of the 42 protocols by Forest Service region. Protocol Veg A, "Ground-Based Skidding and Harvesting," had the highest number of evaluations completed (76), followed by Range A, "Grazing Management" (57), and Fire A, "Use of Prescribed Fire" (41). These three protocols account for 29 percent of all BMP evaluations completed in FY 2014. Only 2 of the 42 BMP monitoring protocols were not used during FY 2014: Road G, "Snow Removal and Snow Storage," and Road I, "Equipment Refueling or Servicing Areas."

Figure 1. Number and percentage of National BMP monitoring evaluations completed in FY 2014, by resource category.



BMP = Best Management Practice. FY = fiscal year.

Note: The resource categories are Aquatic Ecosystems Management Activities (AqEco), Chemical Use Management Activities (Chem), Facilities and Nonrecreation Special Uses Management Activities (Fac), Wildland Fire Management Activities (Fire), Minerals Management Activities (Min), Rangeland Management Activities (Range), Recreation Management Activities (Rec), Road Management Activities (Road), Mechanical Vegetation Management Activities (Veg), Water Uses Management Activities (WatUses).

Table 2. Number of BMP evaluations completed in FY 2014, by Region, for each of the 42 BMP monitoring protocols. (Refer to appendix A for full titles and applications of each of the 42 monitoring protocols.)

Region	AqEco A	AqEco B	Chem A	Chem B	Chem C	Fac A	Fac B	Fac C	Fac D	Fire A	Fire B	Min A	Min B	Min C	Min D	Range A	Rec A	Rec B	Rec C	Rec D	Rec E
R1	2	4	0	0	0	1	0	0	0	2	0	0	0	0	0	8	1	0	1	0	0
R2	1	2	3	0	0	1	1	0	0	2	0	0	0	1	0	6	3	3	0	0	0
R3	0	1	4	0	0	0	1	0	0	6	1	0	0	1	0	5	1	1	1	1	0
R4	1	0	2	0	0	0	0	0	1	2	2	1	1	0	1	13	1	1	2	0	0
R5	1	3	2	0	3	0	1	1	1	3	4	1	0	1	0	9	4	1	1	7	1
R6	5	4	9	0	0	1	3	1	1	8	3	0	4	1	0	8	4	2	2	3	1
R8	5	3	7	1	0	3	2	5	0	11	3	1	0	0	3	6	6	0	3	4	0
R9	3	6	1	1	0	1	0	1	1	7	2	3	0	1	6	2	4	6	4	6	1
R10	1	0	1	0	0	0	2	0	0	0	0	2	0	0	0	0	0	0	1	0	1
Total	**19**	**23**	**29**	**2**	**3**	**7**	**10**	**8**	**4**	**41**	**15**	**8**	**5**	**5**	**10**	**57**	**24**	**14**	**15**	**21**	**4**

Region	Rec F	Rec G	Rec H	Rec I	Road A	Road B	Road C	Road D	Road E	Road F	Road G	Road H	Road I	Veg A	Veg B	Veg C	WatUses A	WatUses B	WatUses C	WatUses D	WatUses E	Total
R1	0	4	1	0	2	1	0	0	1	1	0	1	0	1	0	0	0	0	0	0	1	**32**
R2	1	1	1	0	2	7	3	1	0	0	0	1	0	12	0	3	0	3	0	1	3	**62**
R3	0	0	0	0	0	2	1	0	1	1	0	0	0	6	0	4	1	0	0	0	0	**39**
R4	0	0	0	1	0	4	2	0	1	1	0	1	0	4	0	1	0	3	0	0	1	**47**
R5	2	2	0	3	0	1	5	0	1	2	0	0	0	5	1	5	2	3	2	0	2	**80**
R6	0	2	0	0	4	3	2	5	4	13	0	1	0	14	5	3	0	7	2	0	6	**131**
R8	0	1	0	0	1	5	2	0	0	0	0	1	0	18	0	2	2	1	2	0	0	**98**
R9	1	1	0	0	3	6	5	1	0	2	0	2	0	12	1	2	0	0	1	0	0	**93**
R10	0	0	0	0	0	2	2	1	0	0	0	0	0	4	0	0	0	0	0	1	0	**18**
Total	**4**	**11**	**2**	**4**	**12**	**31**	**22**	**8**	**8**	**20**	**0**	**7**	**0**	**76**	**7**	**20**	**5**	**17**	**7**	**2**	**13**	**600**

BMP = Best Management Practice. FY = fiscal year.

Monitoring Results

Evaluation Rating Outcomes

The purpose of the BMP monitoring rating system is to provide a method of measuring the performance of the Forest Service in applying BMPs and protecting water resources during land management activities on NFS lands. Assigning a rating outcome to each National Core BMP monitoring evaluation will enable tracking of BMP performance over time at multiple scales within the agency. In addition, patterns may emerge that will help to identify strengths and weaknesses in BMP implementation and effectiveness, as well as needed changes in processes or procedures to address identified weaknesses.

For each National Core BMP monitoring evaluation—that is, completion of a monitoring protocol at a selected site—BMP implementation and effectiveness are rated separately. At sites where BMP implementation and effectiveness have both been evaluated, these separate ratings are combined to provide an overall composite BMP performance rating for the site. In this way, BMP implementation and effectiveness can be tracked separately, as can overall BMP performance.

Procedures outlined in the monitoring protocols vary, but the overall approach for each field evaluation is consistent. For BMP implementation, the IDT answers questions to determine whether the activity was executed on the ground as planned in project documents. BMP effectiveness is determined through direct and indirect measures of water resource condition that include observations, measurements, and water quality monitoring data. Scores expressed as ratings for implementation, effectiveness, and composite results are calculated according to protocol-specific rulesets within the BMP database after the data are entered. Appendix B provides a summary of how the rating system is structured and how the rulesets were developed.

The rating categories for implementation are "Fully Implemented," "Mostly Implemented," "Marginally Implemented," "Not Implemented," and "No BMPs." A rating of "No BMPs" is assigned to evaluations that found no evidence that BMPs were included in project planning or in documents that guide operation and maintenance of the site. The primary difference between "Fully Implemented" or "Mostly Implemented" and "Marginally Implemented" is that, in the former two, planned BMPs are implemented fully on the ground, whereas in "Marginally Implemented," some, but not all, planned BMPs are implemented fully on the ground.

The rating categories for effectiveness are “Effective,” “Mostly Effective,” “Marginally Effective,” and “Not Effective.” “Effective” indicates no adverse impacts to water from project or activities were evident. “Mostly Effective” indicates impacts to water resources were minor and temporary. “Marginally Effective” indicates impacts to water resources were minor and prolonged, or major and temporary. Although some protocols incorporate use of existing water quality monitoring data, if available, the protocols do not include direct monitoring of beneficial or designated uses of waterbodies. BMP ratings of “Not Effective” indicate potential for major and prolonged adverse effects to water quality or waterbody condition, but they do not necessarily indicate impairment of beneficial or designated uses.

If a site is selected for BMP evaluation, it is to be assessed first for BMP implementation and then for BMP effectiveness. For most protocols, implementation and effectiveness assessments can be completed in the same day as long as implementation is evaluated first. For those sites where BMP implementation and BMP effectiveness evaluations have both been completed and ratings have been assigned, a composite rating for the evaluation is determined. Appendix B contains the matrix used to determine the composite rating. Composite rating categories are “Excellent,” “Good,” “Fair,” “Poor,” and “No Plan.” The effectiveness rating is given greater weight in the composite rating than the implementation rating, unless the implementation rating was “No BMPs.” If the implementation rating is “No BMPs,” the composite rating is “No Plan” by default because an implementation rating of “No BMPs” represents a failure to consider BMPs in the planning process.

The National BMP monitoring protocols were first used in FY 2013, so during that year, consistent BMP monitoring methodologies became the norm across the agency. The BMP monitoring completed in FY 2013 was used to test the protocols and scoring/rating system; and based on feedback from resource specialists, the protocols and scoring/rating system were significantly revised for FY 2014. Consequently, the FY 2014 BMP evaluations were the first for which scores were calculated and ratings reported.

Of the 600 BMP evaluations competed in FY 2014, 46 (approximately 8 percent) were incomplete; that is, the BMP monitoring database indicated that required information was missing and ratings for either BMP implementation or effectiveness and a composite BMP score could not be calculated. Most of these 46 evaluations were performed using the FY 2013 versions of the protocols, which are not compatible with the revised version of the database, so ratings could not be calculated. The other evaluations, approximately 4 percent of the total evaluations completed, may represent errors in using the monitoring protocol or data entry errors. The evaluations with incomplete data are not included in the rating summary statistics in this report.

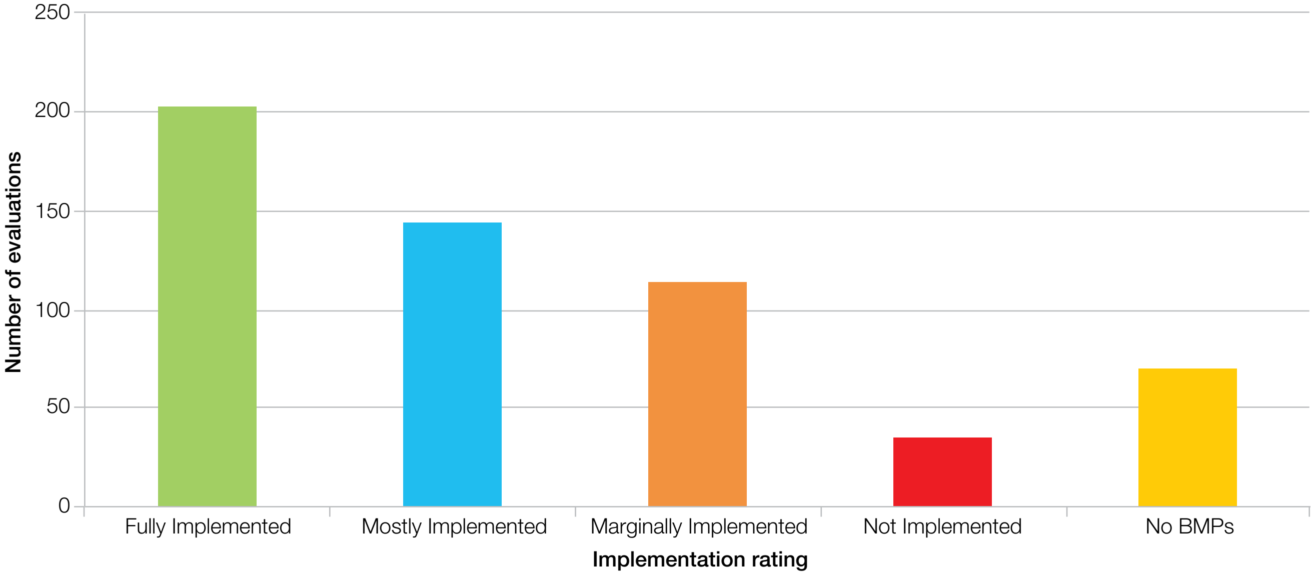

Carry-in boat access to Coffee Lake, Chequamegon-Nicolet National Forest, Wisconsin.

Implementation Ratings

There were 566 evaluations of BMP implementation completed in FY 2014. Figure 2 provides a summary of the BMP implementation ratings for these evaluations. Approximately 35 percent of the evaluations were rated as “Fully Implemented,” 25 percent were rated as “Mostly Implemented,” 20 percent were rated as “Marginally Implemented,” 6 percent were rated “Not Implemented,” and the remaining12 percent were rated as “No BMPs.”

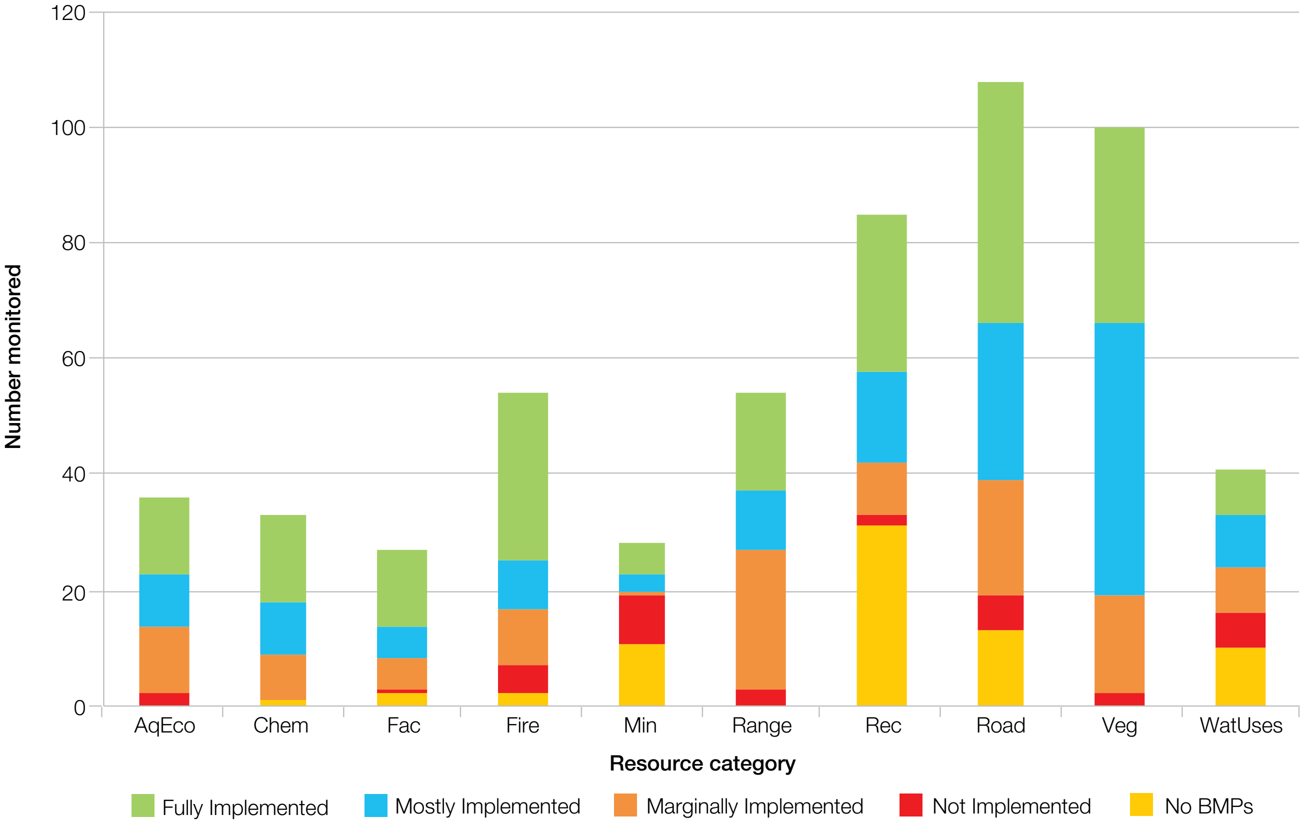
The BMP implementation ratings varied considerably across resource categories (Figure 3). The largest percentage of evaluations rated as “Fully Implemented” was in Wildland Fire Management Activities, with almost 54 percent. Minerals Management Activities has the smallest percentage of evaluations rated as “Fully Implemented,” with slightly less than 18 percent. In 8 of the 10 resource categories, the percentage of evaluations with implementation ratings of “Fully Implemented” or “Mostly Implemented” exceeds 50 percent, led by Mechanical Vegetation Management Activities, with 81 percent. Only Minerals Management Activities and Water Uses Management Activities had less than 50 percent of the evaluations rated as “Fully Implemented” or “Mostly Implemented.” Recreation Management Activities had the largest percentage of evaluations rated as “No BMPs,” with 36 percent.

Figure 2. BMP implementation ratings across all BMP monitoring protocols for evaluations completed in FY 2014.



BMP = Best Management Practice. FY = fiscal year.

Figure 3. BMP implementation ratings, by resource category, for evaluations completed in FY 2014.



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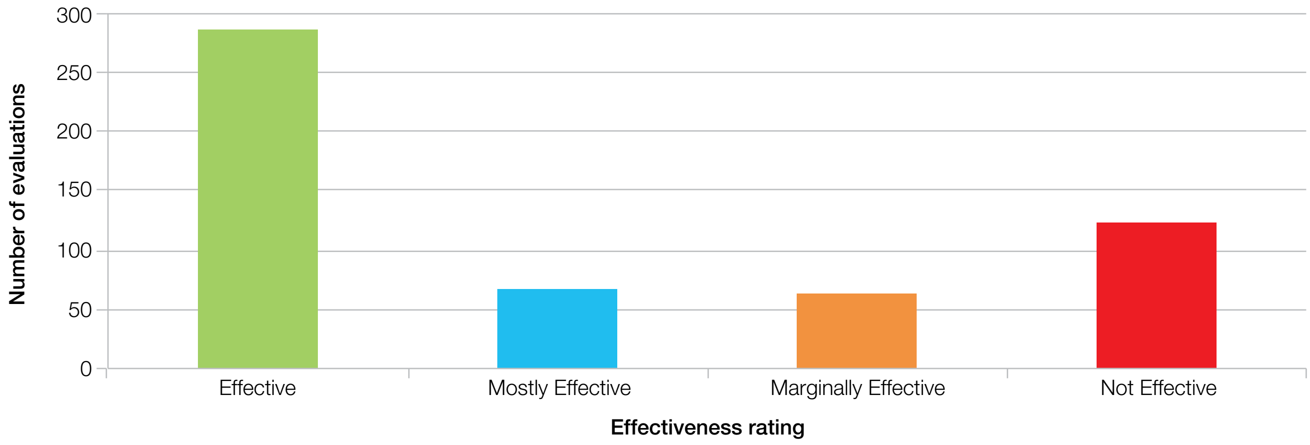
Note: The resource categories are Aquatic Ecosystems Management Activities (AqEco), Chemical Use Management Activities (Chem), Facilities and Nonrecreation Special Uses Management Activities (Fac), Wildland Fire Management Activities (Fire), Minerals Management Activities (Min), Rangeland Management Activities (Range), Recreation Management Activities (Rec), Road Management Activities (Road), Mechanical Vegetation Management Activities (Veg), Water Uses Management Activities (WatUses).

Effectiveness Ratings

There were 539 evaluations of BMP effectiveness completed in FY 2014. Figure 4 provides a summary of the BMP effectiveness ratings for these evaluations. Approximately 53 percent of the evaluations were rated as "Effective," 12 percent were rated as "Mostly Effective," 12 percent were rated as "Marginally Effective," and the remaining 23 percent were rated as "Not Effective."

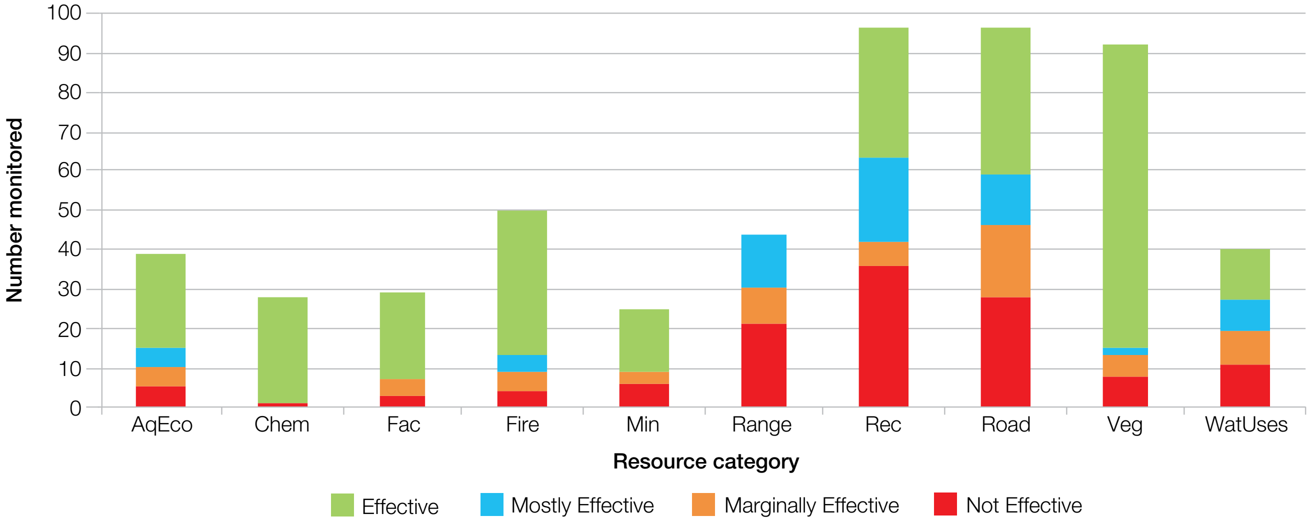
As with the BMP implementation ratings, the BMP effectiveness ratings varied considerably across resource categories (Figure 5). Chemical Use Management Activities had the highest BMP effectiveness rating, at 96 percent, followed by Mechanical Vegetation Management Activities, at almost 84 percent, and Facilities and Nonrecreation Special Uses Management Activities, at nearly 76 percent. The three resource categories with the lowest percentages of evaluations having BMP effectiveness ratings of "Effective" were Rangeland Management Activities, at 0 percent, Water Uses Management Activities, at almost 33 percent, and Recreation Management Activities, at 34 percent. Most of the resource categories, however, had more than 50 percent of the evaluations with BMP effectiveness ratings of "Effective" or "Mostly Effective." Rangeland Management

Figure 4. BMP effectiveness ratings across all BMP monitoring protocols for evaluations completed in FY 2014.



BMP = Best Management Practice. FY = fiscal year.

Figure 5. BMP effectiveness ratings, by resource category, for evaluations completed in FY 2014.



BMP = Best Management Practice. FY = fiscal year.
Note: The resource categories are Aquatic Ecosystems Management Activities (AqEco), Chemical Use Management Activities (Chem), Facilities and Nonrecreation Special Uses Management Activities (Fac), Wildland Fire Management Activities (Fire), Minerals Management Activities (Min), Rangeland Management Activities (Range), Recreation Management Activities (Rec), Road Management Activities (Road), Mechanical Vegetation Management Activities (Veg), Water Uses Management Activities (WatUses).

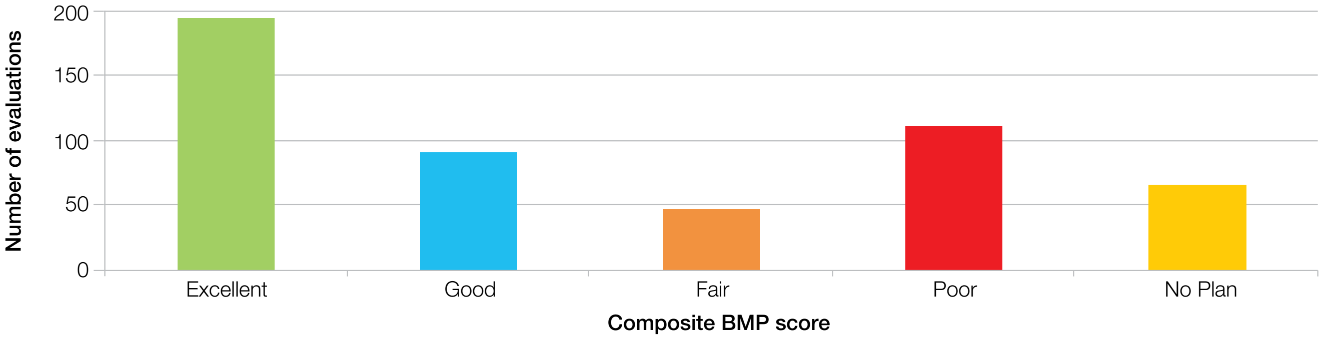
Activities was the exception, with only about 32 percent of the evaluations rated as "Effective" or "Mostly Effective" and the remaining 68 percent rated as "Marginally Effective" or "Not Effective" at achieving water resource objectives.

Composite Ratings

There were 509 evaluations completed in FY 2014 for which a composite rating for BMP implementation and effectiveness could be determined (Figure 6). Composite ratings were "Excellent" for 38 percent of the evaluations, "Good" for 18 percent, "Fair" for 9 percent, and "Poor" for 22 percent. The remaining 13 percent of the evaluations had BMP implementation ratings of "No BMPs" and, therefore, had a composite rating of "No Plan."

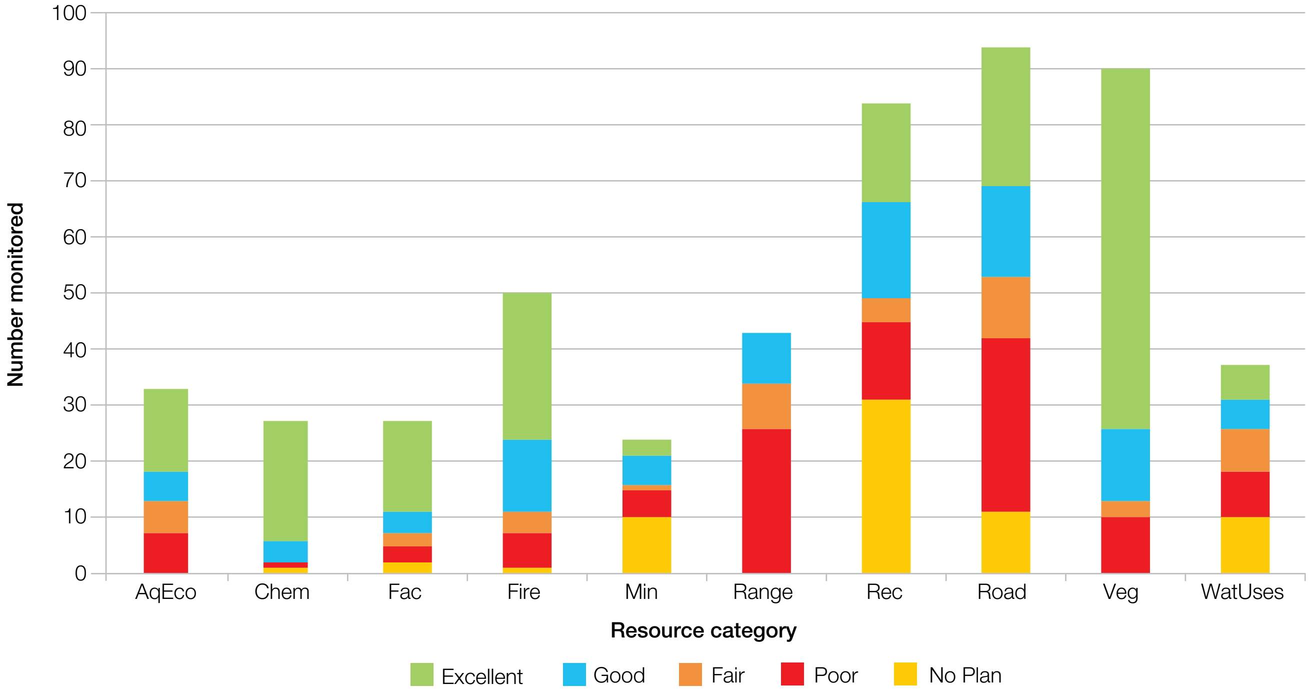
Not surprisingly, based on the implementation and effectiveness ratings, Chemical Use Management Activities and Mechanical Vegetation Management Activities had the highest percentages of evaluations with composite ratings of "Excellent" (Figure 7), with 78 and 71 percent, respectively. The percentage of evaluations with composite ratings of either "Excellent" or "Good" exceeded 70 percent for four resource categories: Chemical Use

Figure 6. Composite BMP evaluation ratings across all BMP monitoring protocols for evaluations completed in FY 2014.



BMP = Best Management Practice. FY = fiscal year.

Figure 7. Composite BMP evaluation ratings, by resource category, for evaluations completed in FY 2014.



BMP = Best Management Practice. FY = fiscal year.

Note: The resource categories are Aquatic Ecosystems Management Activities (AqEco), Chemical Use Management Activities (Chem), Facilities and Nonrecreation Special Uses Management Activities (Fac), Wildland Fire Management Activities (Fire), Minerals Management Activities (Min), Rangeland Management Activities (Range), Recreation Management Activities (Rec), Road Management Activities (Road), Mechanical Vegetation Management Activities (Veg), Water Uses Management Activities (WatUses).

Management Activities, at 93 percent, Mechanical Vegetation Management Activities, at 86 percent, Wildland Fire Management Activities, at 78 percent, and Facilities and Nonrecreation Special Uses Management Activities, at 74 percent. The percentage of evaluations with composite ratings of “Fair,” “Poor,” or “No Plan” exceeded 60 percent for three resource categories: Rangeland Management Activities, at 79 percent, Water Uses Management Activities, at 70 percent, and Minerals Management Activities, at 67 percent. Recreation Management Activities, Road Management Activities, Minerals Management Activities, and Water Uses Management Activities had the highest number of evaluations with composite ratings of “No Plan.”

Appendix C contains summary figures of BMP implementation, BMP effectiveness, and composite ratings by protocol for the BMP evaluations completed in FY 2014.

Corrective Actions and Adaptive Management

BMP assessments provide the opportunity to determine if corrective actions or adaptive management actions are needed for implementation and effectiveness of BMPs. The National BMP monitoring protocols differentiate between corrective actions and adaptive management actions. Corrective actions are typically actions applied to problems identified for or at the project or site being evaluated. Adaptive management actions are actions that typically would be applied broadly to management of all sites, projects, or activities like that one being evaluated.

Corrective actions for implementation are applicable when something that should have been implemented was not. No effectiveness problem needs to exist for an implementation corrective action to be identified or applied; corrective actions identified during the review of implementation simply note that something was supposed to been done but it was not, so there is an opportunity to correct that deficiency. By contrast, corrective actions identified during evaluations of BMP effectiveness generally are associated with an observed problem, because BMPs that were applied were not fully effective. Examples showing the differentiation between corrective actions for implementation and effectiveness are provided in Table 3.

Bottomless pipe-arch crossing on Case Camp Ridge Road, Pisgah National Forest, North Carolina.

Table 3. Examples of corrective actions and adaptive management actions for BMP implementation and BMP effectiveness.

Type of evaluation	Corrective actions	Adaptive management actions
BMP implementation	Return to site and install water control structures that were specified in the contract but were not constructed.	Ensure that expected or acceptable sediment inputs to streams during culvert replacements are described during planning so there is a threshold against which to compare actual inputs.
		Have the NEPA coordinator review all contracts before release to ensure they include ALL of the BMPs from the decision notice.
BMP effectiveness	Fix undersized waterbars on skid roads that have failed or are overtopped by runoff during rain events.	Cease prescribing and using silt fence in all projects or where concentrated flow is present because they are consistently undercut, sidecut, or overtopped.
	Remove and treat soil contaminated by hydraulic fluids from equipment failure during this project.	Change Forest Plan Standards and Guidelines for Aquatic Management Zone widths on side slopes greater than 45 percent to a minimum of 200 feet, as widths of less than that do not allow reinfiltration of emergent flow resulting from cut slope construction.

BMP = Best Management Practice. NEPA = National Environmental Policy Act.

Corrective actions for effectiveness can be characterized as either short- or long-term efforts. Short-term actions typically are those that require little or no additional planning to address water, aquatic, and riparian impacts associated with the project or site, such as fixing a waterbar on a skid trail that is contributing sediment to a stream. In some situations, more substantive actions will provide more sustainable long-term solutions to an observed problem. These actions often will lead to overall improvements in watershed condition or health. For example, rerouting a road adjacent to a stream channel that chronically contributes large sediment inputs to the stream could be a long-term corrective action identified to improve effectiveness. The size, scope, and cost of these more impactful types of corrective actions generally require additional planning. Administrative units also will consider these actions thoroughly to determine if they align with future watershed condition objectives. During BMP evaluations, short- and long-term corrective actions are identified for effectiveness when appropriate, and the corrections are categorized as a short- or long-term action.

Identification of adaptive management actions usually involves observations of recurring problems or common deficiencies over time. As a consequence, in many if not most cases, adaptive management actions are not applied to the current project or site being evaluated but rather to future projects or sites of that type or that have similar attributes, such as all mechanical harvesting operations.

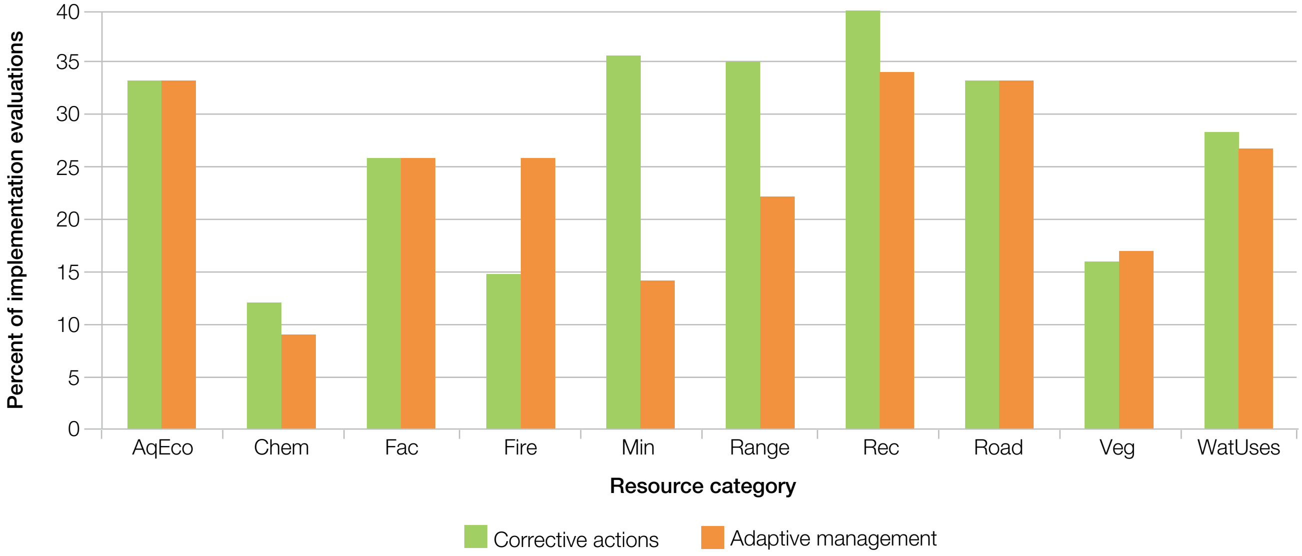
Adaptive management actions for implementation often involve adjustments to processes during planning, such as ensuring plans are written for force account projects to ensure all involved parties have the same understanding and expectations. The lack of BMP effectiveness is still central to adaptive management actions identified for effectiveness, but the actions typically result either from consistently observing, or observing in certain situations, BMPs that work well or poorly. As a result, adaptive management actions may involve a conscious change in how or where certain BMPs are applied in the future. Examples of adaptive management actions for implementation and effectiveness also are provided in Table 3.

The very act of performing implementation and effectiveness monitoring acknowledges uncertainty about the degree to which BMPs are planned and implemented within the agency and BMP efficacy. The feedback loop involved in identifying and applying both corrective and adaptive management actions provides the Forest Service with mechanisms to make adjustments if BMPs are not applied or they are less than fully effective, or to identify situations in which new BMP designs or prescriptions are needed. Undertaking identified corrective actions or adaptive management actions, however, is done at the discretion of the administrative unit's responsible official after considering the risk to water quality, unit work priorities, staffing, funding, and other resource limitations (USDA Forest Service in prep.).

Corrective actions for BMP implementation were identified in approximately 28 percent (161) of the BMP implementation evaluations completed in FY 2014 (Figure 8). A similar number (26 percent; 145) of these evaluations identified adaptive management actions for BMP implementation. As a percentage of BMP implementation evaluations completed, corrective actions (40 percent) and adaptive management actions (34 percent) were identified most often in Recreation Management Activities BMP evaluations and least often in Chemical Use BMP evaluations (12 percent for corrective actions, 9 percent for adaptive management strategies).

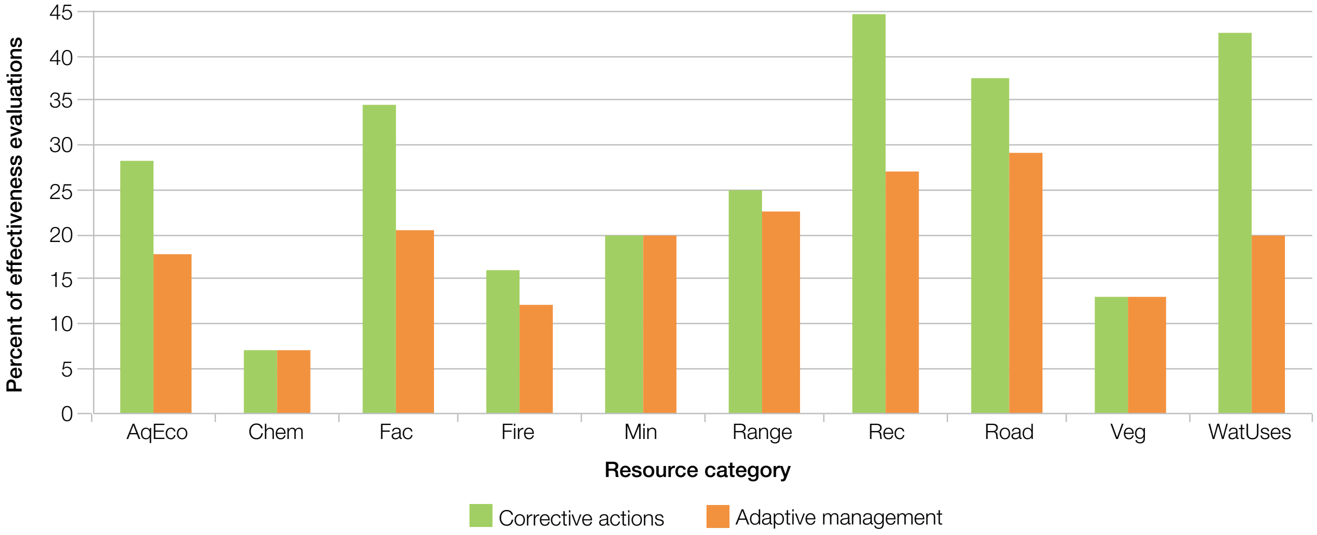
Corrective actions for BMP effectiveness were identified in approximately 29 percent (155) of the BMP effectiveness evaluations completed in FY 2014 (Figure 9). As with implementation evaluations, corrective actions were identified most often in Recreation Management Activities effectiveness evaluations (44 percent) and least often in Chemical Use effectiveness evaluations (7 percent). Adaptive management actions were identified in 20 percent (110) of the FY 2014 BMP effectiveness evaluations. Adaptive management actions were identified most often in Road Management Activities effectiveness evaluations (29 percent) and least often in Chemical Use Management Activities effectiveness evaluations (7 percent).

Figure 8. Percentage of BMP implementation evaluations with identified corrective actions and adaptive management actions, by resource category, for evaluations completed in FY 2014.



BMP = Best Management Practice. FY = fiscal year.
Note: The resource categories are Aquatic Ecosystems Management Activities (AqEco), Chemical Use Management Activities (Chem), Facilities and Nonrecreation Special Uses Management Activities (Fac), Wildland Fire Management Activities (Fire), Minerals Management Activities (Min), Rangeland Management Activities (Range), Recreation Management Activities (Rec), Road Management Activities (Road), Mechanical Vegetation Management Activities (Veg), Water Uses Management Activities (WatUses).

Figure 9. Percentage of BMP effectiveness evaluations with identified corrective actions and adaptive management actions, by resource category, for evaluations completed in FY 2014.



BMP = Best Management Practice. FY = fiscal year.
Note: The resource categories are Aquatic Ecosystems Management Activities (AqEco), Chemical Use Management Activities (Chem), Facilities and Nonrecreation Special Uses Management Activities (Fac), Wildland Fire Management Activities (Fire), Minerals Management Activities (Min), Rangeland Management Activities (Range), Recreation Management Activities (Rec), Road Management Activities (Road), Mechanical Vegetation Management Activities (Veg), Water Uses Management Activities (WatUses).

Interdisciplinary team checking postfire soil conditions, Umatilla National Forest, Oregon.

Discussion

One purpose of the 2-year phase-in period to full implementation of the National BMP Program was to provide time for administrative units to become accustomed to using the National BMP Program tools and procedures. The phase-in approach has been successful in this regard.

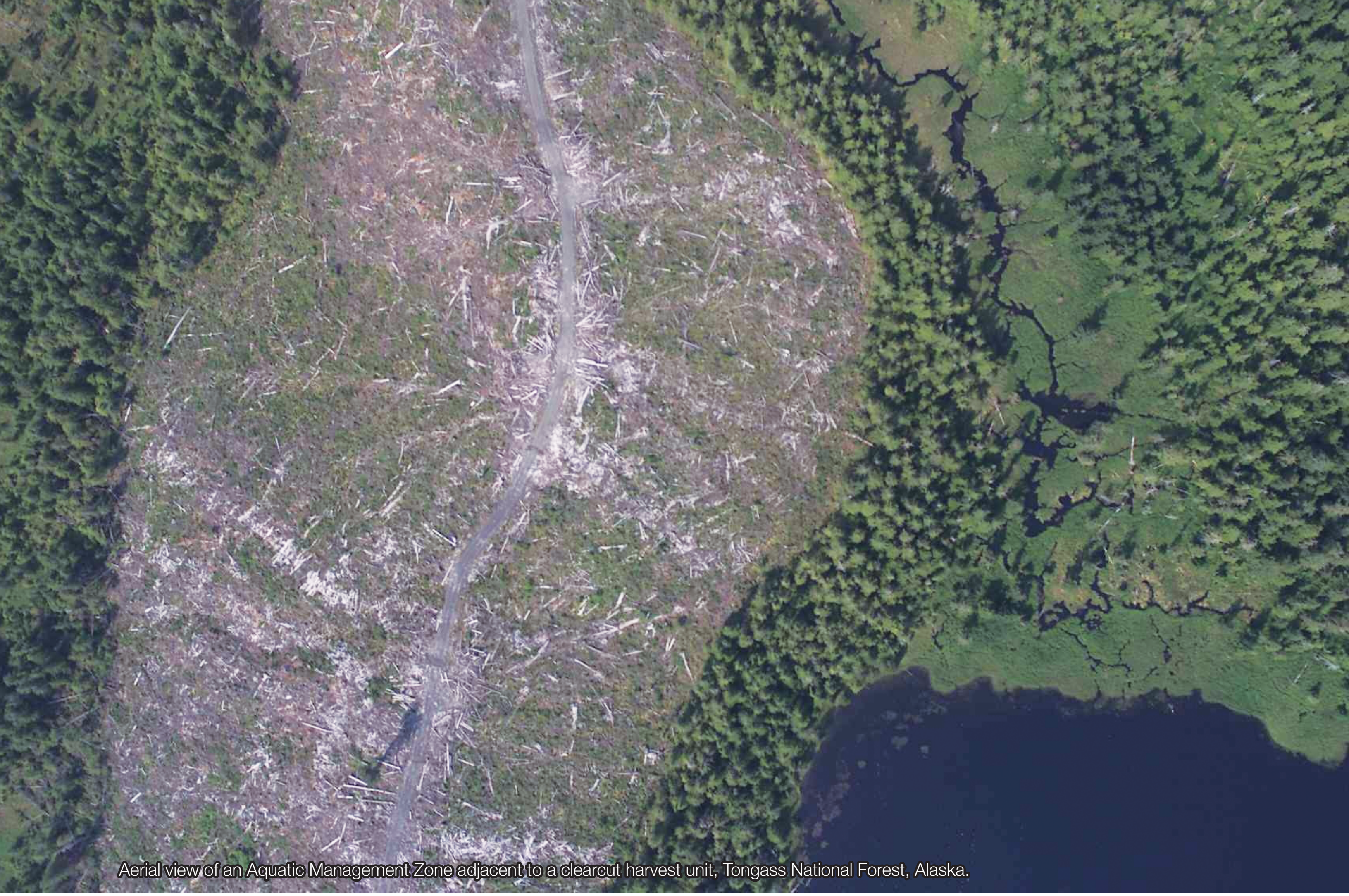
By the end of FY 2014, nearly 90 percent of the administrative units had completed at least one BMP evaluation. This figure is an improvement from FY 2013, when 26 percent of the administrative units did not complete at least one BMP evaluation (USDA Forest Service 2015). Most of the BMP evaluations completed were assessments of the most common activities that occur on NFS lands. In FY 2014, over one-half of the BMP evaluations completed were in the Road Management Activities, Recreation Management Activities, and Mechanical Vegetation Management Activities resource categories. This percentage is similar to FY 2013 BMP monitoring, when 64 percent of the completed evaluations were in these three resource categories. In FY 2014, only two monitoring protocols (Road G and Road I) were not used at least once, also an improvement from FY 2013, when four of the protocols were not used: Chem B, "Chemical Use in Waterbodies," Chem C, "Chemical Use for Dust Abatement," Min A, "Active Construction of Mineral Exploration Sites and Predevelopment Activities (Nonplacer Mining)," and WatUses D, "Active Construction of Diversions and Conveyances." Over the course of the 2-year period, all of the individual monitoring protocols were used somewhere on NFS lands at least once. In addition, in FY 2013, the percentage of completed BMP evaluations with missing data in the database was approximately 8 percent (USDA Forest Service 2015). In FY 2014, this percentage was cut in half, with only 4 percent of the completed evaluations having incomplete data in the database.

The other purpose of the phase-in period was to test and refine the National BMP monitoring protocols and associated rating rulesets. The phase-in approach has been successful in this regard as well. Feedback from resource specialists on FY 2013 monitoring results indicated that the ratings did not reflect their professional assessment of site conditions, BMP implementation, and BMP effectiveness. Feedback on the FY 2014 monitoring results indicates that, with the revisions to the protocols and rating system, the ratings are much closer to the professional judgment of the resource specialists. The only concerns that were expressed were related to the Range A effectiveness ratings, so this rating ruleset will be reviewed again before being finalized.

The phase-in period also demonstrates the potential power of having a National BMP Program. Each Forest Service administrative unit was asked to complete a small number of BMP evaluations in FY 2014, and the result is that the agency has over 500 BMP monitoring data points with which to document BMP implementation and effectiveness. The initial BMP monitoring shows that 61 percent of the BMP implementation evaluations were rated as "Fully Implemented" or "Mostly Implemented," 65 percent of the BMP effectiveness evaluations were rated as "Effective" or "Mostly Effective," and 56 percent of the sites where both BMP implementation and effectiveness were monitored had composite ratings of "Excellent" or "Good." While these data show room for improvement in BMP implementation and effectiveness across the agency, prior to development of the National BMP Program, it was impossible to report on BMP implementation and effectiveness on a national scale in a coherent, understandable, and useful way.

Use of standardized monitoring protocols with rating outcomes also allows for identification of patterns and, eventually, trends in BMP implementation and effectiveness. The FY 2014 monitoring results show that the best overall performances of BMP implementation and BMP effectiveness, as indicated by the percentage of evaluations rated as "Excellent" or "Good," were in the Mechanical Vegetation Management Activities, Chemical Use Management Activities, and Wildland Fire Management Activities resource categories. Mechanical Vegetation Management Activities and Chemical Use Management Activities also had the lowest percentage of evaluations that identified corrective actions or adaptive management actions to improve BMP implementation or effectiveness. These resources have a long history of emphasis on the use of BMPs to protect water quality.

Also not too surprising is the finding that the resource category with the highest number of composite ratings of "No Plan," meaning no BMPs were prescribed, was Recreation Management Activities. Recreation Management Activities also had the highest percentage of evaluations that identified corrective actions or adaptive management actions to improve BMP implementation or effectiveness. Most of the protocols in this resource category assess ongoing operation and maintenance of existing facilities (campgrounds, trails, water launches, etc.), which are guided by operation and maintenance plans. The process for identifying and incorporating appropriate BMP prescriptions into operation and maintenance plans may not be as straightforward as it is



Aerial view of an Aquatic Management Zone adjacent to a clearcut harvest unit, Tongass National Forest, Alaska.

for construction projects or timber sale projects that go through the National Environmental Policy Act, or NEPA, analysis and documentation process and are implemented primarily through contracts.

The identification of corrective actions and adaptive management actions will be most useful at the local administrative unit and regional scales. The results of BMP monitoring, especially the scoring and rating, can be used at the national scale. For example, in resource areas that struggle with low implementation and/or effectiveness outcomes, adaptive management may take the form of increased funding or training in an effort to improve the outcome of those resource activities. The FY 2014 results show that the administrative units are using the monitoring protocols to capture information on how to improve BMP implementation and effectiveness in the future as part of an adaptive management or continuous learning process.

Conclusion

BMP monitoring has been conducted on NFS lands for many years, but there has been little consistency across regions or administrative units in how BMPs were monitored or the data were summarized. The National BMP Program has addressed these shortcomings by providing a nationally consistent, systematic, and objective approach to BMP monitoring, which serves as a foundation for water quality protection on NFS lands (USDA Forest Service 2012).

In FY 2014, some incomplete data reporting occurred on a small number of BMP evaluations, some administrative units used the incorrect versions of the protocols, and some administrative units did no BMP monitoring at all using the national protocols. Incomplete BMP monitoring results cannot be used for national reporting purposes because no ratings can be calculated for BMP implementation or effectiveness in the BMP database when data are missing. Incomplete monitoring efforts do not contribute to national objectives and goals and also do not provide the full set of information that would otherwise be available to the local unit. As a consequence, greater effort must be made to ensure all required information is collected during BMP evaluations and correctly entered into the BMP database. Additional "train-the-trainer" sessions and BMP database training webinars will be held in FY 2015 and beyond to address this issue. In addition, other training possibilities involving a variety of media options are being considered to increase BMP monitoring training opportunities in the future.

During FY 2014, the Forest Service completed the 2-year phase-in period of the National BMP Program. The FY 2014 BMP monitoring results show that the agency is capable of implementing and monitoring BMPs using a national program. As the agency moves from the phase-in period into full implementation of the program in FY 2015, finalization of the protocols and rating system will allow the BMP monitoring results to be used to determine trends in BMP implementation and effectiveness at local, regional, and national scales. As regions and administrative units analyze BMP monitoring results, improvement in BMP implementation and effectiveness is anticipated through (1) improved consistency in field monitoring, (2) the identification of BMP deficiencies and recommendations for corrective and adaptive management actions, and (3) improved BMP planning during project development and operation and maintenance of sites.

The Forest Service is continually monitoring the implementation and effectiveness of BMPs as well as improving methodologies, data storage, management, and reporting. With sustained focus on improving every facet of the BMP program, the agency can ensure greater transparency and long-term protection of water quality and aquatic resources.

Low-flow stream crossing, Black Hills National Forest, South Dakota.

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Spring box with protective fencing, Wallowa-Whitman National Forest, Oregon.

Appendix A
BMP Monitoring Protocols and Descriptions

Resource category	Protocol	Use to evaluate	Examples of appropriate project, activity, or site
General Planning Activities	Planning is addressed in the protocols of every resource category. Specific monitoring protocols do not exist for the General Planning Activities category.		
Aquatic Ecosystems Management Activities	AqEco A Active Construction of Aquatic Ecosystem Improvements	• Aquatic ecosystem improvements during construction or reconstruction • Soil-disturbing improvements in waterbodies • Soil-disturbing improvements in the floodplain	• Fish habitat improvement (excluding road culvert removal for aquatic organism passage—use Road protocol) • Stream restoration • Bank stabilization • Wetland construction
	AqEco B Completed Aquatic Ecosystem Improvements	• Completed aquatic ecosystem improvement projects in the floodplain • Completed aquatic ecosystem projects applied to a waterbody	• Fish habitat improvement (excluding road culvert removal for aquatic organism passage—use Road protocol) • Stream restoration • Bank stabilization • Wetland construction
Chemical Use Management Activities	Chem A Chemical Use Near Waterbodies	• Chemical use near waterbodies where the target or objective was terrestrial • Aerial applications of chemicals with terrestrial targets, even if no attempt was made to discontinue application over waterbodies	• Control of terrestrial noxious weeds • Chemical silvicultural treatments • Fertilizer and lime applications to improve soil nutrition/chemistry
	Chem B Chemical Use in Waterbodies	• Chemicals applied to waterbodies where the target or objective was an aquatic species or water chemistry	• Chemical control of aquatic invasive species • Stream liming
	Chem C Chemical Use for Dust Abatement	• Use of road dust abatement chemicals, excluding water-only applications	• Applications of any type of dust palliative
Facilities and Nonrecreation Special Uses Management Activities	Fac A Active Construction of Noncorridor Facilities or Nonrecreation Special Uses	• Completed construction of noncorridor types of facilities administered by the Forest Service • Completed construction of nonrecreation facilities administered by special use permits (SUP)	Construction and reconstruction of: • Campgrounds • Ski area base facilities • Concessions operated under special use authorization • Communications facilities • Water treatment facilities • Forest Service administrative facilities • Grazing units or pastures authorized under special use authorizations *other than* Grazing Permits with Term Status
	Fac B Operation and Maintenance of Noncorridor Facilities or Nonrecreation Special Uses	• Operation and maintenance of noncorridor types of facilities administered by the Forest Service • Operation and maintenance of nonrecreation facilities administered by special use permits (SUP)	Operation and maintenance of: • Ski area base facilities • Communications facilities • Water treatment facilities • Forest Service administrative facilities • Grazing units or pastures authorized under special use authorizations *other than* Grazing Permits with Term Status

Resource category	Protocol	Use to evaluate	Examples of appropriate project, activity, or site
Facilities and Nonrecreation Special Uses Management Activities (continued)	Fac C Completed Construction or Operation and Maintenance of Pipelines, Transmission Lines, or Rights-of-Way	• Completed construction of pipelines, transmission lines, and nonroad rights-of-way • Operation and maintenance of pipelines, transmission lines, and nonroad rights-of-way	• Construction of energy pipelines or transmission lines • Construction of water pipelines that are not associated with diversions • Operation and maintenance of transmission lines, energy pipelines, and water pipelines that are not associated with diversions
	Fac D Completed Facility Reclamation	• Completed reclamation of facilities, including recreation facilities, administrative sites, structures, and pipelines and transmission lines (i.e., nonroad corridors)	Reclamation of: • Sites that held residences, historic structures, or other buildings • Areas previously occupied by ski areas, campgrounds, or concentrated-use areas • Transmission line and pipeline corridors • Trails or trail segments that will no longer be used
Wildland Fire Management Activities	Fire A Use of Prescribed Fire	• Planning and implementation of prescribed fires	• Prescribed fire for any purpose
	Fire B Wildfire Management Actions	• Management of monitored fires • Management of suppressed fires	• Wildfires for which the management action taken is not suppression • Actively suppressed fires • Fires that have locations or periods involving both suppression and nonsuppression (monitoring)
Rangeland Management Activities	Range A Grazing Management	• Grazing and livestock management under a Grazing Permit with Term Status	• Permitted grazing of livestock and any associated range improvements (e.g., stock pond construction and maintenance and fencing)
Minerals Management Activities	Min A Active Construction of Mineral Exploration Sites and Predevelopment Activities (Nonplacer Mining)	• Construction at nonplacer mineral sites to prepare for exploration • Predevelopment activities at nonplacer minerals sites to prepare for production	• Construction or predevelopment activities for minerals outside of waterbodies and alluvial deposits (i.e., in the AMZ) • Includes hard rock, solid leasable minerals, coal mining, oil and gas sites, geothermal activities and other minerals
	Min B Active Nonplacer Mineral Operations	• Exploration operations and active mineral operations that do not involve placer mining	• Exploration and active mineral operations involving hard rock, metallic minerals, coal mining, phosphate mining, oil and gas sites, and other minerals, excluding extraction of minerals from waterbodies or alluvial deposits in the AMZ
	Min C Placer Mining Operations	• Placer mining for any type of mineral; includes extraction from the waterbody or AMZ (i.e., in alluvium)	• Placer mining operation authorized by an Approved Plan of Operations or negotiated terms • Includes suction dredging, locatable minerals, or sand and gravel mining extracted from the waterbody or from alluvial deposits in the AMZ
	Min D Reclamation of Mineral Operations	• Reclamation of construction and predevelopment disturbances • Reclamation of exploration sites where no further development or extraction occurred • Reclamation of all types of placer and nonplacer mineral operations	Reclamation activities of: • Hard rock, metallic mineral, coal, and phosphate mines • Oil and gas well pads • Suction dredging sites, sand and gravel operations, gold mining, and other mining operations in a waterbody or in an AMZ • Improvements and disturbances associated with the mining or extraction, as well as additional land disturbances created to complete reclamation

Resource category	Protocol	Use to evaluate	Examples of appropriate project, activity, or site
Recreation Management Activities	Rec A Developed Recreation Sites	• Operation and maintenance of developed recreation sites	Operation and maintenance of: • Campgrounds • Day-use areas, including picnicking, swimming, rock climbing, or fishing areas
	Rec B Dispersed Recreation Areas	• Dispersed-use recreation	• Undeveloped camping areas • Undeveloped picnicking, swimming, rock climbing, or fishing areas • High-use undeveloped areas that may or may not have sanitary facilities or trash facilities
	Rec C Completed Construction or Rerouting of Motorized or Nonmotorized Trails	• Construction or rerouting of Forest Service-authorized motorized trails • Construction or rerouting of Forest Service-authorized nonmotorized trails	• Construction of new trails • Construction to extend existing trails • Rerouting of trail segments to move trails away from waterbodies or to overlooks
	Rec D Motorized or Nonmotorized Trail Operation and Maintenance	• Operation and maintenance of Forest Service-authorized motorized trails • Operation and maintenance of Forest Service-authorized nonmotorized trails	• Use of existing system trails Maintenance of existing system trails that may or may not involve soil disturbance, including: • Removal of downed trees on trails • Repair or reconstruction of handrails on high-use trails • Repair or replacement of water control features • Replacement of logs as crossing structures over streams
	Rec E Motorized Vehicle Use Areas	• Operation and maintenance of motor vehicle use areas designated for off-highway vehicles (OHVs, ATVs, 4-wheel drive trucks, dune buggies, etc.)	• OHV trails located within a motor vehicle use area • Motor vehicle use areas containing concentrated use areas such as mud holes, mud bogs, or hill climbs
	Rec F Pack and Riding Stock Use Areas	• Operation and maintenance of pack and riding stock use areas	• Corrals or similar holding areas, and stock watering areas
	Rec G Active Construction or Operation and Maintenance of Watercraft Launches	• Active construction of watercraft launches • Operation and maintenance of watercraft launches	• Construction of boat ramps, launches, and marinas • Operation of boat ramps, launches, and marinas • Maintenance of existing watercraft launches • Use of backcountry canoe and kayak launches
	Rec H Completed Ski Area Construction or Reconstruction	• Ski area construction or reconstruction	• Construction of ski runs, lift lines, or snowmaking systems involving vegetation clearing or ground disturbance • Ground disturbance at on-hill ski facilities from construction of lift towers or other support structures and utilities
	Rec I Ski Run Operation and Maintenance	• Ski run operation and maintenance in which soil is not disturbed substantially	• Ski run use • Routine maintenance of ski runs, including mowing during the offseason and snow grooming during the ski season
Road Management Activities	Road A Active Road or Waterbody Crossing Construction or Reconstruction	• Active road or waterbody crossing construction or reconstruction • Includes work on Forest Service system roads, as well as work on nonsystem roads and crossings authorized by road use agreements, special use permits, or minerals plans of operation when Forest Service has significant input into planning, BMP implementation and project supervision	• Road construction or reconstruction • Construction or reconstruction of waterbody crossings even if other road work is not being performed • Removal or replacement of waterbody crossing structures on roads to improve aquatic organism passage • Active reconstruction treatments to prepare a road for storage

Resource category	Protocol	Use to evaluate	Examples of appropriate project, activity, or site
Road Management Activities (continued)	Road B Completed Road or Waterbody Crossing Construction or Reconstruction	• Completed road or waterbody crossing construction or reconstruction • Includes work on Forest Service system roads, as well as work on nonsystem roads authorized by road use agreements, special use permits, or minerals plans of operation when Forest Service had significant input into planning, BMP implementation and project supervision	• Constructed or reconstructed roads • Constructed or reconstructed waterbody crossings even if other road work was not performed • Completed removal or replacement of waterbody crossing structures on roads to improve aquatic organism passage
	Road C Road Operation and Maintenance	• Long-term management and maintenance of Forest Service maintenance level 2 through 5 system roads	• Road use of both gated and open roads • Routine road maintenance (e.g., road grading or resurfacing)
	Road D Stored Roads	• Forest Service system roads that are currently designated as maintenance level 1 roads	• Forest Service system roads or road segments that have been placed into storage because they are not needed for long periods
	Road E Active Road Decommissioning	• Active road decommissioning projects • Decommissioning of Forest Service system roads of any maintenance level, as well as nonsystem roads originally authorized by road use agreements, special use permits, or minerals plans of operation • Includes off-forest roads as long as the Forest Service is responsible for implementing BMPs and project supervision	Activities employed during road decommissioning, including but not limited to: • Removing waterbody crossing structures • Restoring hillside drainage patterns • Stabilizing slopes and restoring vegetation • Spreading slash on road surface • Road obliteration by restoring natural hillside slopes and contours • Blocking road entrances
	Road F Completed Road Decommissioning	• Completed road decommissioning activities • Decommissioning of Forest Service system roads of any maintenance level, as well as nonsystem roads originally authorized by road use agreements, special use permits, or minerals plans of operation • Includes off-forest roads as long as the Forest Service is responsible for implementing BMPs and project supervision	Roads decommissioned by a variety of practices, including but not limited to: • Removing waterbody crossing structures • Restoring hillside drainage patterns • Stabilizing slopes and restoring vegetation • Spreading slash on road surface • Road obliteration by restoring natural hillside slopes and contours • Blocking road entrances
	Road G Snow Removal and Snow Storage	• Snow removal from Forest Service system roads of any maintenance level • Snow removal from parking areas when associated with road snow removal • Snow storage areas associated with snow removed from evaluated road	• Snow removal by plowing, blowing, mechanically lifting and moving, or deicing • Stored snow removed from parking areas
	Road H Completed Construction/ Reconstruction or Operation and Maintenance of Parking Areas	• Construction/reconstruction of permanent parking areas • Use of permanent parking areas • Maintenance of permanent parking areas	• Parking lot construction or reconstruction • Use of parking areas • Maintenance of parking area surfacing and drainage • Maintenance of oil and grease containment or separator systems • Includes parking areas for administrative areas, developed recreation sites, visitor centers, trail heads, roadside rests, and scenic overlooks
	Road I Equipment Refueling or Servicing Areas	• Designated temporary equipment service at active project sites • Temporary refueling areas designated to store at least 1,320 gallons of oil and fuels at active project sites	• Areas designated for heavy equipment repair and maintenance within timber harvest units • Refueling areas designated at road construction projects

Resource category	Protocol	Use to evaluate	Examples of appropriate project, activity, or site
Mechanical Vegetation Management Activities	Veg A Ground-Based Skidding and Harvesting	• Completed ground-based skidding and harvesting operations	• Typical timber harvesting operations involving log skidding and temporary storage of logs on landings • Ground-based timber/vegetation removal for facility development including recreation sites, ski areas, campgrounds, administrative sites, or road construction
	Veg B Cable and Aerial Yarding Operations	• Completed harvesting in which log transport was by cable or other aerial yarding system	• Felling followed by cable transport of logs along corridors • Helicopter logging
	Veg C Mechanical Site Treatments	• Completed mechanical site treatments	• Site preparation, such as chopping residual vegetation using heavy equipment • Vegetation pile burning as part of other site preparation activities • Timber stand improvement treatments using chainsaws or heavy equipment • Mechanical control or removal of terrestrial invasive species • Fuels reduction treatments using chainsaws or heavy equipment
Water Uses Management Activities	WatUses A Completed Construction or Operation and Maintenance of Water Wells for Monitoring or Production	• Completed construction of water wells to produce water or monitor groundwater levels or condition • Operation and maintenance of existing water wells used to provide water or monitor groundwater levels or condition	• Nested wells at different depths or individual wells for groundwater monitoring studies • Water wells for public use at developed campgrounds • Water wells for administrative facilities
	WatUses B Operation and Maintenance of Spring-Source Facilities	• Operation and maintenance of developed springs	• Water sources fed by springs at campgrounds or roadside rests • Spring sources for livestock watering
	WatUses C Completed Reconstruction/ Repair or Operation and Maintenance of Water Sources (Drafting)	• Completed repair or reconstruction of water drafting sources • Operation and maintenance of existing water drafting sources	• Improvements made to water drafting sites • Water drafting sites used for fire suppression, mineral operations, or road dust control
	WatUses D Active Construction of Diversions and Conveyances	• Construction and reconstruction of permanent water diversion and/or water conveyance systems, including water storage facilities, temporary access roads or staging areas for the project, return flow	• Diversion or conveyance systems for range management or irrigation • Diversion or conveyance systems authorized by special use permit
	WatUses E Operation and Maintenance of Diversions and Conveyances	• Operation and routine maintenance of existing permanent diversions, conveyances, and associated water storage and return flow	• Operation of diversion and conveyance facilities used for range management or irrigation • Operation of conveyance systems authorized by special use permit • Routine maintenance of diversion and conveyance facilities, including sediment or debris removal from the system

AMZ = Aquatic Management Zone. ATV = all-terrain vehicle. BMP = Best Management Practice. OHV = off-highway vehicle.

Best Management Practices training participants discuss possible corrective actions at a recently decommissioned day-use recreation site, Uinta-Wasatch-Cache National Forest, Utah.

Appendix B
BMP Evaluation Rating Rule Set Development

The purpose of the Best Management Practice (BMP) monitoring rating system is to provide a method of measuring the performance of the Forest Service in applying BMPs and protecting water quality during land management activities on National Forest System (NFS) lands. Assigning a rating outcome to each National Core BMP monitoring evaluation will enable tracking of BMP performance over time at multiple scales within the agency. In addition, patterns may emerge that will help to identify strengths and weaknesses in BMP implementation and effectiveness and needed changes in processes or procedures to address identified weaknesses.

In devising the rating system, the following statements of fact were considered:

- Each National Core BMP monitoring protocol evaluates more than one National Core BMP, typically a planning BMP and one or more resource-category BMPs.
- The National Core BMP monitoring protocols are written in general, nonspecific terms and are designed to evaluate BMP performance by assessing outcomes of BMP implementation regardless of the site-specific BMP prescription used.
- The protocol questions are structured so as to obtain objective information on BMP implementation and effectiveness at a site.
- The BMP evaluations will be completed by an interdisciplinary review team (IDT) of professional resource specialists.
- Water quality impacts are inferred from visual evidence of pollutant movement offsite and into nearby waterbodies, changes to waterbody morphology, and, where available, existing water quality or other relevant monitoring data.
- Water chemistry, habitat quality, and other water quality parameters are not measured directly. While some protocol questions concern water quality standards, there is no attempt to quantify attainment of water quality standards.
- A team of people from the Washington Office and regional offices decided the rating outcome categories and definitions.

Therefore, the National Core BMP monitoring protocols are designed to use a qualitative assessment by knowledgeable professionals to evaluate overall BMP implementation and effectiveness for an activity, such as developed recreation or road construction, being monitored. They are not designed to be a quantitative evaluation of site-specific BMP prescriptions or individual National Core BMPs.

For each National Core BMP monitoring evaluation—that is, completion of a monitoring protocol at a selected site—BMP implementation and effectiveness are rated separately. These separate ratings are combined to provide an overall BMP performance rating for the site. In this way, BMP implementation and effectiveness can be tracked separately, as can overall BMP performance.

The ratings for BMP implementation and effectiveness are determined, based the combination of answer choices selected in the BMP evaluation, according to a ruleset developed individually for each National Core BMP monitoring protocol. Routines consistent with the ruleset within the BMP-monitoring-data management system will analyze the answers to the protocol questions and assign the ratings to each evaluation. The monitoring IDT will not assign the site ratings directly. In addition, the number of questions in the various protocols and weighting applied to the various protocol answer choices should make it difficult for the IDT to "rig" the answers to achieve a specific or better rating outcome.

BMP Implementation Rating Outcomes

BMP implementation monitoring answers the question, "Were site-specific BMP prescriptions implemented as planned or designed?" This question has two parts: (1) "What site-specific BMP prescriptions were planned or designed?" and (2) "Were the site-specific BMP prescriptions implemented as intended?"

Planning establishes "What site-specific BMP prescriptions were planned or designed?" Monitoring of planning includes review of project planning documents, such as a project Environmental Impact Statement and associated Record of Decision, or other guidance documents, such as the land management plan or State BMPs, to identify site-specific BMP prescriptions. Monitoring of planning also includes review of project-implementing documents to determine if those planned BMP prescriptions were included in project contracts, permits, or other implementing documents.

Operational execution of planning addresses the question, "Were those site-specific BMP prescriptions implemented as intended?" Monitoring of operational execution involves a field review of the project area to determine if the specified BMP prescriptions were implemented and if corrective actions were taken if problems with those specified prescriptions or other water quality-related issues were identified during the course of the project or activity. Implementation questions in the National BMP monitoring protocols are designed to obtain information about BMP planning and operational execution. The implementation rating categories are shown in Table B-1.

Note the implementation rating is based solely on the BMP prescriptions included in the planning or guidance documents or project implementation documents. This evaluation does not answer the question of what BMPs should have been prescribed, which is often clearer in hindsight than in the planning phase. There is the opportunity, however, to provide comments on this issue if it is found that planning was inadequate or not appropriate. Also note, the rating category "No BMPs" represents a total failure of the BMP process in planning and is distinguished from "Not Implemented," in which BMPs were identified in planning but not included in action documents or implemented fully.

To determine the implementation rating, selected implementation questions in each protocol are divided into three groups:

- BMPs were prescribed (BMPs Rx)
- BMPs were implemented (BMPs Imp)
- Corrective actions were implemented (C.A.)

Each group of implementation questions is given a rating of "All," "Some," or "None" based on the combination of their answer choices. The three group ratings are then combined into the implementation rating.

BMPs were prescribed (BMPs Rx): This grouping of implementation questions addresses planning, or "What site-specific BMP prescriptions were planned or designed?" In most protocols, the BMPs Rx rating is based on two types of implementation questions: (1) "What is the planning document or other BMP guidance document?" and (2) "Were the BMP provisions in the those documents included in the project implementation document?" Some protocols have additional implementation questions that address site-specific BMP prescriptions that are also factored into the BMPs Rx rating. For example, protocol Road C asks if Road Management Objectives (RMOs) were established for the road and if those RMOs reflect existing design and use.

Table B-1. Definitions of rating categories for BMP implementation.

Implementation rating	Interpretation
Fully Implemented	Prescriptions are identified in project planning documents, –and– All prescriptions are translated into action documents, –and– All specified prescriptions are implemented fully, –and– All necessary corrective actions identified during the project are implemented fully.
Mostly Implemented	Prescriptions are identified in project planning documents, –and– All or Some prescriptions are translated into action documents, –and– All specified prescriptions are implemented fully, –and– All or Some necessary corrective actions identified during the project are implemented fully.
Marginally Implemented	Prescriptions are identified in project planning documents, –and– All or Some prescriptions are translated into action documents, –and– Some specified prescriptions are implemented fully, –and– All or Some necessary corrective actions identified during the project are implemented fully.
Not Implemented	Prescriptions are identified in project planning documents, –and– No prescriptions are translated into action documents, –or– No specified prescriptions are implemented fully, –or– No necessary corrective actions identified during the project are implemented.
No BMPs	Site-specific BMP prescriptions were not developed or identified during project planning.

BMP = Best Management Practice.

The BMPs Rx rating is “All,” “Some,” or “None” depending on the degree to which BMP prescriptions were established for the project or activity and were included in the project implementing document. If there is no planning or other BMP guidance document, the BMPs Rx rating is “No BMPs.” If there is a planning or other BMP guidance document but it does not contain site-specific BMP prescriptions, the BMPs Rx rating is “No BMPs Rx.”

BMPs were implemented (BMPs Imp): This grouping of implementation questions addresses operational execution of planning, or “Were the site-specific BMP prescriptions implemented as intended?” In most protocols, the BMPs Imp rating is based on one comprehensive implementation question about which BMP provisions in the project implementing document were implemented fully on the ground. Some protocols have additional implementation questions that address implementation of site-specific BMP prescriptions, which are also factored into the BMPs Imp rating. For example, protocol WatUses A asks if the well apron or collar meets all State and local requirements for materials, size, and thickness.

The BMPs Imp rating is “All,” “Some,” or “None” depending on the degree to which BMP prescriptions were implemented fully at the site.

Corrective actions were implemented (C.A.): This grouping of implementation questions also addresses operational execution and looks at whether water quality problems are recognized and corrected during project implementation or ongoing activities. The types of implementation questions used for the C.A. rating include questions about whether inspections of the project site were made, if supplemental erosion control was needed, and if the site was closed or improvement treatments were applied. The C.A. rating is “All,” “Some,” or “None” depending on the degree to which water quality problems were identified and corrected during project implementation.

Implementation rating: The BMPs Rx, BMPs Imp, and C.A. ratings are combined into the implementation rating for an evaluation, as shown in Table B-2.

BMP Effectiveness Rating Outcomes

BMP effectiveness monitoring answers the question, “Were the site-specific BMP prescriptions, as implemented, effective at protecting water quality?” Effectiveness monitoring assesses the prevention of pollutants from moving into a waterbody and prevention of adverse effects to a waterbody. Pollutant movement and potential threat are judged by how many occurrences and the type of visible evidence of pollutants attributable to the project or activity being evaluated are found in the Aquatic Management Zone (AMZ) or waterbody. “Adverse effects to a waterbody” refers to negative physical disturbance or other change to waterbody morphology from the project or activity being evaluated. Effectiveness questions in the monitoring protocols are designed to obtain information about the presence and movement of pollutants offsite and observable disturbances to a waterbody. The effectiveness rating categories are shown in Table B-3.

To determine the effectiveness rating, selected effectiveness questions are divided into groups of related questions, typically by pollutant type or location at the project site. For example, in a particular protocol, all questions about erosion and sedimentation may be grouped together in one group and questions about trash and sanitary waste are placed in a separate group. For another, in a different protocol, all erosion and sedimentation questions pertaining to the AMZ may be in one group and all erosion and sedimentation questions pertaining to the waterbody crossing may be placed in a separate group. The number of groups in each protocol depends on the number of effectiveness questions and how they are organized. Some protocols have as few as 2 effectiveness groups, and more complicated protocols may have as many as 12. A Group effectiveness rating is assigned to each grouping of effectiveness questions. Depending on the nature of the questions and the ability to distinguish effects from the questions asked, the group effectiveness rating is either a three-category scale or four-category scale. The three-category scale is “Effective,” “Moderately Effective,” and “Not Effective.” The four-category scale is

Table B-2. Matrix to determine the implementation rating.

Implementation rating	BMPs Rx	BMPs Imp	C.A.
Fully Implemented (all are true)	All	All	All
Mostly Implemented (all are true)	Some	All	All or some
	All	All	Some
Marginally Implemented (all are true)	All or some	Some	All or some
Not Implemented (any one is true)	None	None	None
No BMPs (either is true)	No BMPs or no BMPs Rx		

BMP = Best Management Practice.

Table B-3. Definitions of rating categories for BMP effectiveness.

Effectiveness rating	Interpretation	
Effective	No pollutants reached the waterbody and there is no potential threat evident, –and– Waterbody received no adverse effects from the project or activity (e.g., physical disturbance).	
Mostly Effective	Minor amounts of pollutants reached the waterbody or there is a potential threat evident, –and/or– Waterbody received minor adverse effects from the project or activity, –and/or– Impacts to water quality are temporary, lasting less than 1 year.	
Marginally Effective	Minor amounts of pollutants reached the waterbody or there is a potential threat evident, –and/or– Waterbody received minor adverse effects from the project or activity, –and/or– Impacts to water quality are prolonged, lasting more than 1 year.	Major amounts of pollutants reached the waterbody or there is a potential threat evident, –and/or– Waterbody received major adverse effects from the project or activity, –and/or– Impacts to water quality are temporary, lasting less than 1 year.
Not Effective	Major amounts of pollutants reached the waterbody or are very close to entering the waterbody, –or– Waterbody received major adverse effects from the project or activity, –and– Impacts to water quality are prolonged, lasting more than 1 year.	

BMP = Best Management Practice.

"Effective," "Mostly Effective," "Marginally Effective," and "Not Effective." The group ratings are then combined to determine the overall effectiveness rating for the evaluation.

To determine the Group effectiveness rating, the answer choices for each question within the group are rated as "Not Applicable," "Effective," "Mostly Effective," "Moderately Effective," "Marginally Effective," "Not Effective," "No Potential Threat," "Potential Threat," or "Major Potential Threat." For example, the answer choice "no evidence of erosion or sedimentation" is rated as "Effective," whereas the answer choice "flow was poorly controlled or uncontrolled" is rated as "Potential Threat." Each group has at least one effectiveness question, and some can have five or more. The Group effectiveness rating is generally based on the worst rating of the answer choices selected within that grouping. That is, generally all questions in the group need to be rated as "Effective" or "No Potential Threat" in order for the Group effectiveness rating to be "Effective." If any of the questions within the group are rated as "Not Effective," the Group effectiveness rating is also "Not Effective."

The overall effectiveness rating for the evaluation is also based on the worst rating of the group ratings in that evaluation. In order for the overall effectiveness rating to be "Effective," all the group ratings have to be "Effective." If any of the group ratings are "Not Effective," the overall effectiveness rating is "Not Effective" as well. An overall effectiveness rating of "Mostly Effective" results when at least one of the group ratings is "Mostly Effective" and none are "Marginally Effective" or "Not Effective." For example, Table B-4 shows the criteria for the overall effectiveness rating for protocol Road E.

Table B-4. Matrix to determine the effectiveness rating for protocol Road E, active road decommissioning.

Road E effectiveness rating	ER[WBC][a]	ER[RS][b]	ER[CF][c]
Effective (all are true)	Effective –or– no waterbody crossing	Effective	Effective
Mostly Effective	Any combination of **ER[WBC]** or **ER[RS]** is Mostly Effective or **ER[CF]** is Moderately Effective, –and– neither **ER[WBC]** or **ER[RS]** is Marginally Effective, –and– none of **ER[WBC]**, **ER[RS]**, or **ER[CF]** is Not Effective.		
Marginally Effective	Any combination of **ER[WBC]** or **ER[RS]** is Marginally Effective, –and– none of **ER[WBC]**, **ER[RS]**, or **ER[CF]** is Not Effective.		
Not Effective (any are true)	Not Effective	Not Effective	Not Effective

[a] ER[WBC] is effectiveness rating for waterbody crossing.
[b] ER[RS] is effectiveness rating for road segment.
[c] ER[CF] is effectiveness rating for chemicals and fuels.

Overall BMP Performance Rating Outcomes

Once the evaluation ratings for BMP implementation and effectiveness have been decided, an overall BMP performance rating for that BMP evaluation will be determined according to the matrix in Table B-5. There are five possible overall BMP performance ratings: “Excellent,” “Good,” “Fair,” “Poor,” and “No Plan.” In determining the overall performance rating, greater weight is given to the effectiveness rating. For example, an overall rating of “Excellent” can be achieved even if the implementation rating is not “Fully Implemented” as long as the effectiveness rating is “Effective.” The overall performance rating of “No Plan” is assigned to an evaluation for which the implementation rating is “No BMPs,” which means that no BMPs were prescribed for the project or activity during planning. This rating represents a total failure of the BMP process and is a negative outcome, even if the effectiveness rating is “Effective.”

Table B-5. Matrix for determining overall BMP performance rating for a site evaluation.

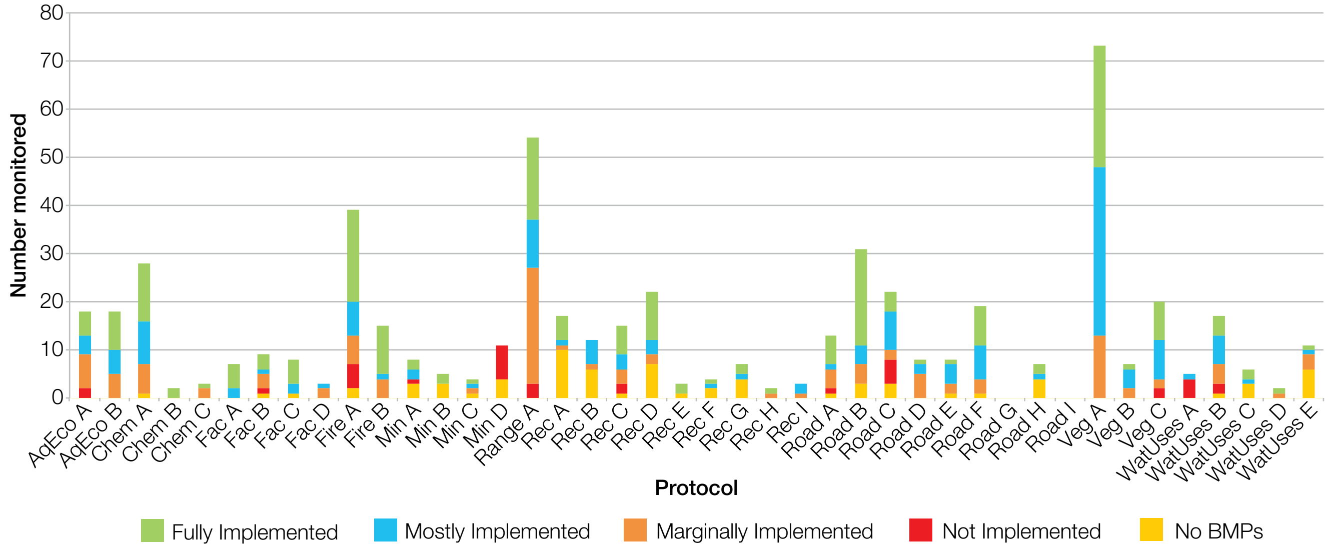
Combined scoring		Implementation rating (IR)				
		Fully Implemented	**Mostly Implemented**	**Marginally Implemented**	**Not Implemented**	**No BMPs**
Effectiveness rating (ER)	**Effective**	Excellent	Excellent	Good	Good	No Plan
	Mostly Effective	Good	Good	Fair	Fair	No Plan
	Marginally Effective	Fair	Fair	Poor	Poor	No Plan
	Not Effective	Poor	Poor	Poor	Poor	No Plan

BMP = Best Management Practice.

Best Management Practice (BMP) training participants discuss BMP effectiveness at a log landing in a commercial timber sale, Coconino National Forest, Arizona.

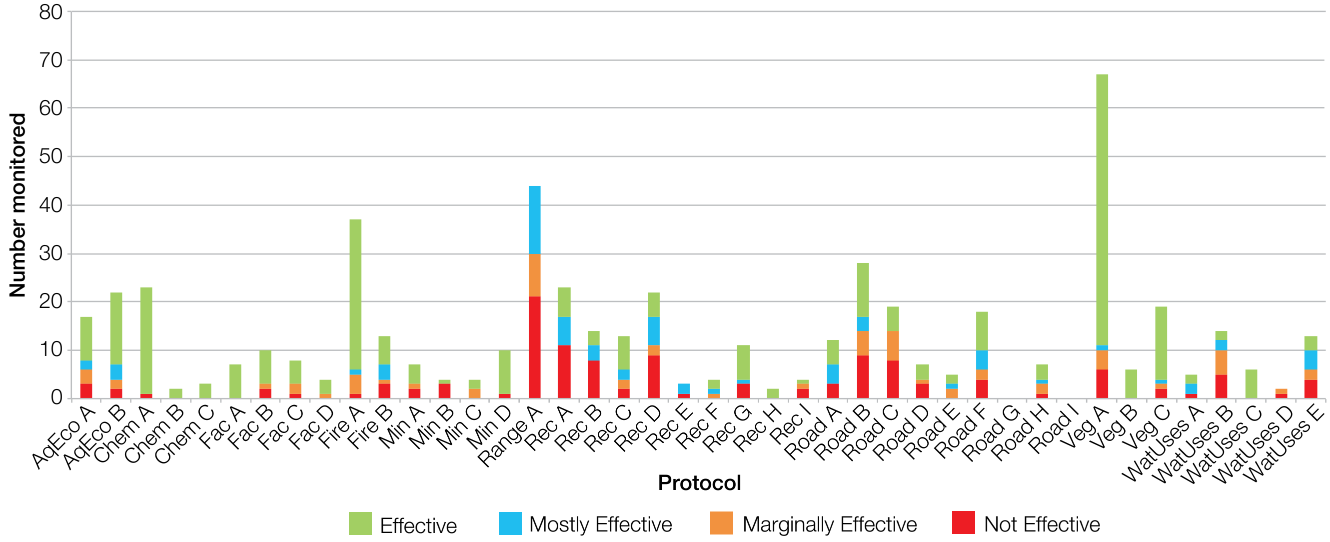
Appendix C
BMP Evaluation Ratings by Protocol

Figure C-1. BMP implementation ratings, by protocol, for evaluations completed in FY 2014.



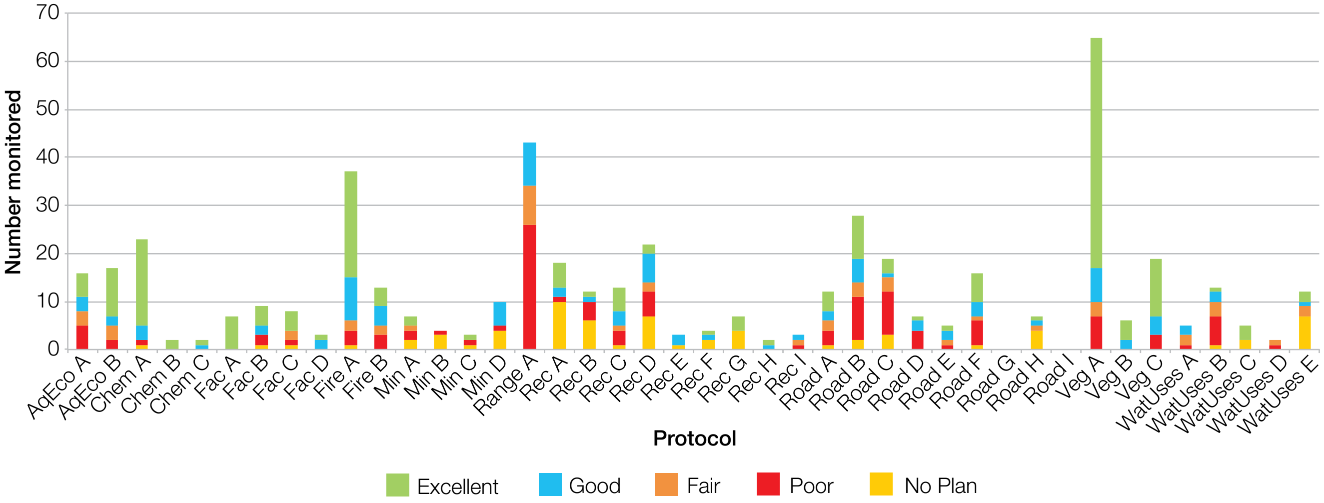
BMP = Best Management Practice. FY = fiscal year.
Note: Appendix A includes protocol explanations.

Figure C-2. BMP effectiveness ratings, by protocol, for evaluations completed in FY 2014.



BMP = Best Management Practice. FY = fiscal year.
Note: Appendix A includes protocol explanations.

Figure C-3. Composite BMP evaluation ratings, by protocol, for evaluations completed in FY 2014.



BMP = Best Management Practice. FY = fiscal year.
Note: Appendix A includes protocol explanations.

sustainability

MDPI

Article

Terrestrial Condition Assessment for National Forests of the USDA Forest Service in the Continental US

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Abstract: The terrestrial condition assessment (TCA) evaluates effects of uncharacteristic stressors and disturbance agents on land-type associations (LTAs) to identify restoration opportunities on national forest system (NFS) lands in the United States. A team of agency scientists and managers, representing a broad array of natural resource disciplines, developed a logic structure for the TCA to identify appropriate data sources to support analyses. Primary national data sources included observed insect- and pathogen-induced mortality, key critical loads for soil and the atmosphere, long term seasonal departures in temperature and precipitation, road densities, uncharacteristic wildfires, historical fire regime departure, wildfire potential, insect and pathogen risk, and vegetation departure from natural range of variability. The TCA was implemented with the ecosystem management decision support (EMDS) system, a spatial decision support system for landscape analysis and planning. EMDS uses logic models to interpret data, synthesizes information over successive layers of logic topics, and draws inferences about the ecological integrity of LTAs as an initial step to identifying high priority LTAs for landscape restoration on NFS lands. Results from the analysis showed that about 74 percent of NFS lands had moderate or better overall ecological integrity. Major impacts to ecological integrity included risk of mortality due to insects and disease, extent of current mortality, extent of areas with high and very high wildfire hazard potential, uncharacteristically severe wildfire, and elevated temperatures. In the discussion, we consider implications for agency performance reporting on restoration activities, and subsequent possible steps, including strategic and tactical planning for restoration. The objective of the paper is to describe the TCA framework with results from a national scale application on NFS lands.

Keywords: ecological integrity; stressors; disturbance agents; spatial decision support; restoration; assessment

1. Introduction

National forests and grasslands, under the management of the U.S. Department of Agriculture Forest Service (USFS), have been experiencing unprecedented impacts due to uncharacteristic stressors and disturbance agents over the past few decades. The U.S. burns twice as many acres as three decades ago [1], fire seasons on average have been extended by 78 days in the western United States [2], and the largest insect and disease infestation on record globally is occurring in the western United States and Canada [3]. Multiple stressors are responsible for these problems, in particular warming temperatures, over-stocking and altered fuel complexes in fire dependent ecosystems due to fire suppression, and invasive species.

The USFS has conducted restoration related activities for decades, however the need for reestablishing and retaining resilience of national forest system (NFS) lands to achieve sustainable management has never been greater. The imminent risk of insects and disease, uncharacteristically high rates of mortality that have already occurred, and the extensive areas with high or very high wildfire hazard potential are major concerns of the USFS. Deleterious effects of elevated temperature and reduced precipitation, particularly in the west, uncharacteristically severe or frequent wildfire, fragmentation of habitat due to roads, and the effects of air pollution or invasive species are also adversely impacting NFS lands.

As a consequence, the USFS has made restoration a major priority within the agency. Policy [4], collaborative landscape restoration projects, and on-the-ground activities have emerged in response to restoration needs, with 1.9 million ha treated for restoration needs in 2014 alone. The team conducting the terrestrial condition assessment (TCA) was commissioned to develop a comprehensive assessment of resource conditions and stressors that may warrant restoration consideration to assist in identifying terrestrial restoration opportunities and improve the agency's transparency and accountability for terrestrial restoration investments. The TCA was chartered by the sustainable land management board of directors, composed of Washington office leadership from NFS, research, and state and private forestry branches of the USFS.

The TCA was designed to complement the watershed condition framework (WCF), a national effort to evaluate the status of watersheds across all NFS lands [5]. The TCA and WCF share goals of assessing resource conditions, but the focus and approach differ. The WCF focuses on conditions and stressors affecting water quality and quantity, and aquatic organisms and their habitat, uses watersheds as analytical and reporting units, and is based primarily on expert opinion in a paneling process that scores indicators of the ecological integrity of watersheds. The TCA addresses terrestrial outcomes, uses landscape-scale analytical and reporting units, is data-driven with existing national data sets, and provides an assessment of ecological integrity based on data interpretation and analyses.

The TCA assesses conditions and processes affecting the ecological integrity of landscape ecosystems on NFS lands. The concept of ecological integrity has evolved over the years [6–9]. It is commonly accepted that an ecosystem has integrity when its dominant ecological characteristics (composition, structure, function) occur within their natural ranges of variation, and can withstand and recover from perturbations caused by natural environmental processes or human activities [9–11]. Thus, the key elements of ecological integrity should include intactness (in terms of natural ranges of variation of all key indicators), biodiversity and species viability, ecosystem structure, ecological processes, and stressors.

In North America, ecological integrity has been mapped across national parks in Canada by the Canadian park service [9]. The Canadian approach includes ecological, species diversity, and human development measures, organized into biodiversity, ecosystem processes, and stressor categories. The National Park Service and NatureServe have developed a preliminary ecological integrity assessment framework intended to introduce concepts and methods to managers and to highlight their potential use [12]. The system recommends use of NatureServe's ecological systems as a coarse filter of biodiversity, but also employs measures of vulnerable species assemblages and their habitats, and species-level measures of the vulnerability of individual plant and animal species. Threats and stressors including human development, resource extraction, roads, pollution, and climate change are included in the assessment. The TCA estimates the ecological integrity of landscape ecosystems by comparing current conditions and processes to reference conditions and processes, but also includes indicators of uncharacteristic biological and environmental stressors, including air pollution and road density.

The TCA is a mid-scale evaluation of conditions and stressors occurring across NFS lands, utilizing the landtype association tier of the national hierarchical framework of Ecological Units [13] as analysis units. Landtype associations represent the landscape-level units in the hierarchy, averaging 8000 hectares in size. An ultimate goal is to understand the resilience of landscape ecosystems to stressors, as well as the extent and magnitude of various stressors themselves. However, methods and data for quantifying resilience are lacking [14]. Given our present inability to measure ecological resilience, we use estimates of ecological integrity as a proxy and as a means of addressing escalating degradative ecological changes. The primary goals of the TCA are to assist land managers in identifying restoration needs at a national scale, and provide the tools necessary for regional and local applications including science delivery, data access, and guidance on analytical procedures. Secondary goals are to support restoration prioritization activities, and provide a baseline from which restoration and maintenance activities can be tracked and effects on ecological integrity documented. The objective of the paper is to describe the TCA framework with results from a national scale application on NFS lands.

2. Materials and Methods

A team of scientists and resource specialists from NFS, Research and Development, and State and Private Forestry branches of the USFS conceived and designed the TCA. The team addressed questions related to current restoration investments, resource conditions warranting investments, appropriate scale and units of analysis, selection of measureable indicators, data availability and acquisition, and computational methods.

2.1. Study Area and Analysis Units

The TCA included all administrative units (National Forests and National Grasslands) of the NFS of the USFS in the continental United States (Figure 1). The total land area of the NFS is distributed across 112 administrative units (labeled "Forests" in the figure), and covers approximately 86 million ha.

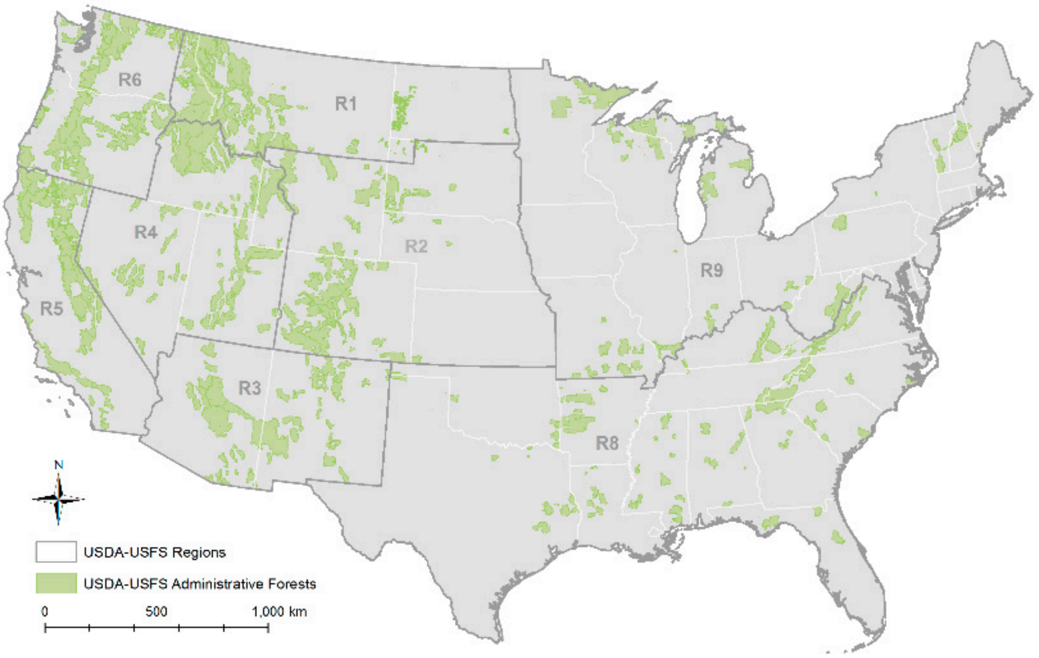


Figure 1. Study area of the Terrestrial Condition Assessment in the continental United States.

Landscape units used in TCA were a combination of landtype associations or generalizations of LANDFIRE's biophysical settings. LTAs are the landscape-level units in the national hierarchical framework of ecological units [13], and are based on patterns in surficial or bedrock geology, lithology, topography, soils and vegetation. LTAs were used in the analysis when these were available for an NFS region. Otherwise, the generalized biophysical settings were used as a close approximation to LTAs.

The objective of using LTAs was to reduce the variability primarily in dominant vegetation as well as natural disturbance regimes. The study area included a total of 10,213 such landscape units. Hereafter, we refer to the analysis units as LTAs.

2.2. Data Sources

Data supporting the TCA were drawn from a variety of sources (Table 1). Estimates for each LTA were derived by zonal statistics using the appropriate input raster indicator dataset and LTA. This methodology was chosen specifically to reduce the overall variance within each estimate but also account for different resolution input indicator datasets. Detailed metadata on the metrics supporting each indicator are included in the supplementary materials .

Table 1. Data sources for metrics used in indicators of the Terrestrial Condition Assessment (TCA).

Indicator	Metric [1]	Data Source
Tree mortality	Mortality due to Insects and Pathogens Data unit: Binary of presence or absence (Ordinal) TCA metric: percent area	National forest pest conditions database produced by USFS forest health technology enterprise team (FHTET) https://foresthealth.fs.usda.gov/portal Raster data at the resolution of 240 m
Terrestrial invasive species	Local Data; Occurrence	NRIS TESP Data are incomplete and not available yet, so this data source is not included in the analysis, although the model includes a placeholder for it.
Road density	Highway road density Paved road density Light duty road Density Unimproved road density Data unit: mi/sq. mi. (Numeric) TCA metric: mi/sq. mi. (Numeric)	USFS FSTOPO transportation dataset developed by USFS geospatial technology and applications center (GTAC) http://data.fs.usda.gov/geodata/vector/index.php Vector line features
Climate exposure	**Temperature**: Mean seasonal temperatures Spring, summer, fall, winter Data unit: Degrees Fahrenheit (Numeric) TCA metric: Degrees F difference **Precipitation**: Total seasonal precipitations Spring, summer, fall, winter % precipitations Spring, summer, fall, winter Data unit: Inches (Numeric) TCA metric: Inches difference	PRISM Climatological Data produced by PRISM Climate Group of Oregon State University with **P**arameter elevation **R**egression on **I**ndependent **S**lopes **M**odel http://prism.oregonstate.edu/ Raster data mostly at the resolution of 4 km
Air pollution	Terrestrial acidification (Exceedance, CAL); Data unit: Ranks of good, moderate, or poor (Ordinal) TCA metric: Ranks of good, moderate, or poor Terrestrial eutrophication (N) Data unit: kg/ha/yr (Numeric) TCA metric: kg/ha/yr (Numeric)	Terrestrial acidification database produced by USFS southern global change program, using the simple mass balance equation (SMBE) http://fsweb.wo.fs.fed.us/wfw/airquality/criticalloads.html Raster data are at the resolution of 1 km^2 Terrestrial eutrophication database generated by EPA's community multiscale air quality (CMAQ) modeling system https://www.epa.gov/air-research/community-multi-scale-air-quality-cmaq-modeling-system-air-quality-management Raster data are at the resolution of 120 m (resampled from 12 km)

Table 1. *Cont.*

Indicator	Metric [1]	Data Source
Catastrophic disturbance	Uncharacteristic fire severity Uncharacteristic fire frequency Data unit: Binary of uncharacteristic and other (Ordinal) TCA metric: Percent area	Database of uncharacteristically severe wildfires derived from (1) Monitoring trends in burn severity (MTBS) data by USGS and USFS and (2) LANDFIRE data of percent low severity fire and percent mixed-severity Fire http://mtbs.gov https://landfire.gov/fireregime.php Raster data at the resolution of 30 m Database of uncharacteristically frequent fire derived from a combination of (1) MTBS as the current condition and (2) Mean fire return interval (MFRI) of LANDFIRE as the reference condition.
Wildfire potential	Uncharacteristic fuel buildup Data unit: Binary of high risk or other TCA metric: Percent area	Wildfire hazard potential (WHP) database produced by USFS Fire Modeling Institute http://www.firelab.org/project/wildfire-hazard-potential Raster data at the resolution of 270 m
Insect and pathogen risk	Potential uncharacteristic mortality Data unit: Binary of presence or absence (Ordinal) TCA metric: Percent area	National insect and disease risk map (NIDRM) produced by USFS forest health protection (FHP) http://www.fs.fed.us/foresthealth/technology/nidrm. shtmlRaster data at the resolution of 270 m
Vegetation departure	Vegetation departure index Data unit: 0–100% (Numeric) TCA metric: Mean	Vegetation departure index (VDEP) produced by LANDFIRE http://www.landfire.gov Raster data at the resolution of 30 m
Ecological process departure	Missed Fire Cycle TCA metric: Mean	Mean fire return interval (MFRI) produced by LANDFIRE http://www.landfire.gov

[1] Metrics represent measurable quantities. Indicators may have one or more metrics.

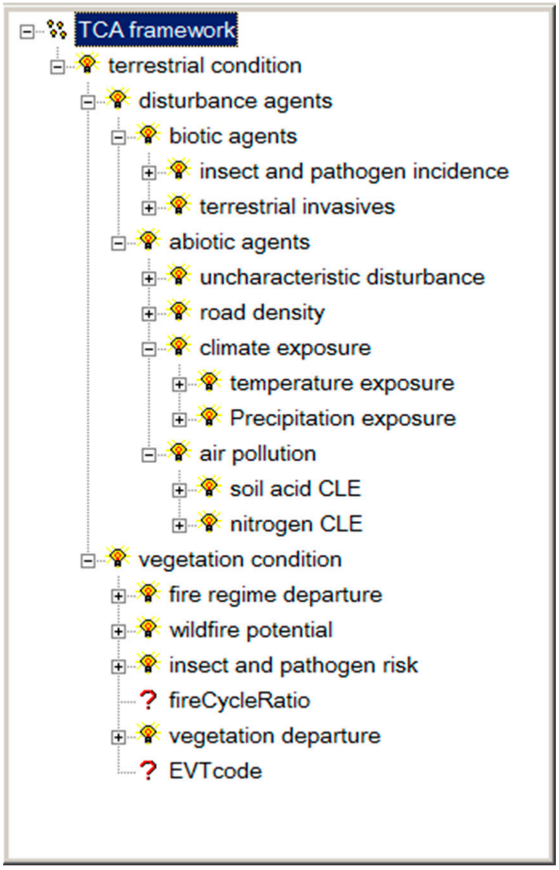
2.3. Overview of EMDS Framework

EMDS is a spatially enabled decision-support framework for integrated landscape evaluation and planning [15]. We describe EMDS as a decision support framework because the data sources, scales of analysis, and models employed in EMDS applications are all user defined. As a result, the system has been applied to a wide variety of decision support problems since 1997 [16,17]. At version 5.5, the system provides decision support for landscape-level analyses through logic and decision engines integrated with the ArcGIS® 10.x geographic information system (GIS, Environmental Systems Research Institute, Redlands, CA, USA), as well as QGIS [18] and MapWindow [19]. The NetWeaver logic engine (Rules of Thumb, Inc., North East, PA) evaluates landscape data against a formal logic specification (e.g., a knowledge base in the strict sense) designed in NetWeaver Developer® [20], to derive logic-based interpretations of ecosystem conditions such as ecosystem integrity. EMDS 5.5 implements the decision engines of three decision support applications. Criterium DecisionPlus® (CDP, InfoHarvest, Seattle, WA, USA) implements the analytical hierarchy process (AHP) [21,22], and can be used for both strategic and tactical planning. GeNIe® (BayesFusion, LLC, Pittsburg, PA, USA) implements Bayesian networks and influence diagrams, while VisiRule® (Logic Programming Associates, Ltd, London, UK) implements Prolog-based decision trees. Both GeNIe and VisiRule are perhaps most applicable to tactical planning in the EMDS context, although strategic applications are also possible. The terms strategic and tactical planning have various interpretations, depending on context. In the particular context of spatial decision support, strategic planning in EMDS is concerned with which management units are the highest priority for management activities, whereas tactical planning is concerned with selecting the highest priority management actions in specific landscape features.

In the present study, our analysis is limited to logic-based processing to assess ecological integrity of LTAs. However, we have introduced the decision engines in this section because their functionality is pertinent to subsequent steps in the larger decision support process that is considered in the later Discussion section.

2.4. NetWeaver Logic Design for TCA

NetWeaver models are implemented as a network of networks (Figure 2). For example, evaluation of the logic network, terrestrial condition, is directly dependent on the evaluation of the two networks, disturbance agents and vegetation condition, at the next lower level of the network outline. Conversely, we can describe the relation as disturbance agents and vegetation condition are logically antecedent to terrestrial condition. Similarly, biotic agents and abiotic agents are the direct logical antecedents of disturbance agents.



- TCA framework
  - terrestrial condition
    - disturbance agents
      - biotic agents
        - insect and pathogen incidence
        - terrestrial invasives
      - abiotic agents
        - uncharacteristic disturbance
        - road density
        - climate exposure
          - temperature exposure
          - Precipitation exposure
        - air pollution
          - soil acid CLE
          - nitrogen CLE
    - vegetation condition
      - fire regime departure
      - wildfire potential
      - insect and pathogen risk
      - ? fireCycleRatio
      - vegetation departure
      - ? EVTcode

Figure 2. Outline of NetWeaver logic structure for the Terrestrial Condition Assessment. The top level, TCA framework, is simply a container for logic networks. Each item in the outline is a logic network, except fireCycleRatio and EVTcode, which are data inputs used to control the flow of logic processing under the network, vegetation condition. Networks listed under vegetation condition represent the latter's logical antecedents (e.g., the evaluation of vegetation conditions depends on fire regime departure, etc.). See the accompanying text for additional explanation of network concepts in NetWeaver.

Apart from its name, each logic network that makes up a logic model has four other important attributes. Each network:

1. Evaluates a proposition about the topic represented by the network, which is contained in a comment field;
2. Has a logical specification composed of its immediate logical antecedents and one or more logic operators that determine how the antecedents contribute to the proposition;
3. Has a measure of the strength of evidence for the proposition provided by its antecedents;
4. Has one or more documentation attributes that describe important aspects of the network (e.g., most networks have an explanation attribute at a minimum).

In the interest of space, the TCA logic outline (Figure 2) is not shown fully expanded. In particular, the model includes additional levels of logic under the networks such as uncharacteristic disturbance, road density, and climate exposure. Comprehensive HTML documentation on the NetWeaver logic for TCA can be found in the supplementary materials section below.

As suggested by the phrase, "a network of networks", NetWeaver models are structurally recursive, such that higher level networks are composed of antecedent networks. This structural recursion terminates with elementary networks that evaluate data. Within each elementary network, data are interpreted with fuzzy logic by comparing observed data values to fuzzy membership functions that translate the observed value into a measure of the strength of evidence for the parent elementary network [20]. Metrics for strength of evidence are propagated upward through the logic structure. Within each network, the strength of evidence metrics contributed by the antecedent networks in the logic specification of their dependent network are logically synthesized by fuzzy logic operators such as AND, OR, and Union [20].

Threshold values presented in Table 2 were established based on a review of the literature, consultation with subject matter experts, and examination of data distributions to ensure model sensitivity. For example, thresholds for indicator 1, extent of insect and disease caused mortality in the past five years, are set at 5% for full evidence of high integrity and 25% for no evidence of high integrity. The 2013–2027 national insect and disease forest risk assessment [23] uses a natural annual background rate of 0.89% for evaluating mortality at a national scale. Based on this literature, the TCA evaluated mortality occurring within the past five years, and considered rates of 5% or less to be natural. The 25% or greater value represents systems that are experiencing mortality at five or more times the natural background rate, affecting ten percent of LTA's nationally. Values between 5% and 25% are ramped and evaluated continuously between these thresholds, such that 7% mortality is very close to full evidence and 23% very close to no evidence.

For readers who may not be familiar with fuzzy logic theory, here, we provide a brief comparison to probability theory, and related issues around confidence limits. Whereas probability theory is concerned with uncertainty in the sense of uncertainty about the likelihood of events, fuzzy logic is concerned with a fundamentally different concept of uncertainty, referred to as linguistic (or lexical) uncertainty [24–26], which originates in the imprecision of human thought and communication. For example, the concept of a warm day is linguistically imprecise (hence fuzzy). Fuzzy logic, or more generally fuzzy math and fuzzy set theory, is actually a precise mathematics for handling imprecise information [24–26]. Fuzzy membership functions, introduced above, are a way of expressing linguistic uncertainty in terms of set theory (e.g., to what degree is an observation a member of some fuzzy set?). The metric for strength of evidence, discussed in the context of NetWeaver, is simply another way of describing degree of set membership, and thus uncertainty. Finally, most readers will have had some training in probabilistic uncertainty, so there is an expectation that our results should include confidence limits on the fuzzy metrics presented in maps of the Results section. There are two compelling reasons why confidence limits are not treated in NetWeaver outputs in EMDS. First, conceptually, doing so would conflate two fundamentally different measures of uncertainty (e.g., probabilistic and linguistic). Second, as a practical matter, the computation of confidence intervals would require solving a convolution integral [27] for the roughly 40 inputs on each of about 10,000 observations, and this assumes that one has error estimates for each of the 400,000 observations, and that the computational algorithm could account for the nonlinearities intrinsic to the logic at runtime. This last reason is a compelling counterargument.

Table 2. Indicators, metrics, and thresholds values used in the Terrestrial Condition Assessment.

Indicator Number	TCA Indicator	Associated Metrics	Threshold for No Evidence [1]	Threshold for Full Evidence [2]	Unit
1	Tree mortality	Mortality due to insects and pathogens	25.0	5.0	% Land-type associations (LTA) area
3	Road density	Highway	0.3	0.1	mile/square mile
		Paved roads	0.3	0.1	mile/square mile
		Light duty roads	1.5	0.5	mile/square mile
		Unimproved roads	2.5	1.0	mile/square mile
4	Climate exposure	Spring temperature	2.0	0.0	°F changed
		Summer temperature	2.0	0.0	°F changed
		Fall temperature	2.0	0.0	°F changed
		Winter temperature	2.0	0.0	°F changed
		Spring precipitation	−1.0	0.0	inch changed
		Summer precipitation	−1.0	0.0	inch changed
		Fall precipitation	−1.0	0.0	inch changed
		Winter precipitation	−1.0	0.0	inch changed
		Spring precipitation (%)	−10.0	0.0	% changed
		Summer precipitation (%)	−10.0	0.0	% changed
		Fall precipitation (%)	−10.0	0.0	% changed
		Winter precipitation (%)	−10.0	0.0	% changed
5	Air pollution	Terrestrial acidification	poor	good	rank
		Terrestrial eutrophication (N)	10.0	1.6	kg/ha/yr
6	Catastrophic disturbance	Uncharacteristic fire severity	5.0	0.0	% LTA area
		Uncharacteristic fire frequency	1.0	1.5	dimensionless
7	Wildfire potential	Uncharacteristic fuel buildup	66.0	20.0	% LTA area
8	Insect and pathogen risk	Potential uncharacteristic mortality	50.0	10.0	% LTA area
9	Vegetation departure	Vegetation departure index	67.0	43.0	% area departed
10	Ecological process departure	Missed fire cycle	35.0	200.0	year departed

[1] Value at which the fuzzy membership function interpreting the associated metric provides no evidence for a suitable condition; [2] Value at which the fuzzy membership function interpreting the associated metric provides full evidence for a suitable condition.

2.5. TCA Analysis in EMDS System

The TCA analysis to assess the ecological integrity of LTAs on NFS lands was implemented in the ArcMap (ESRI) version of the EMDS system. All metrics needed for the assessment (Table 2) were initially obtained or developed as separate GIS layers. Zonal statistics procedures, available in ArcMap, were used to attribute each metric to the LTA polygons. The TCA analysis for the full set of 10,213 LTAs in the continental U.S. was performed with the NetWeaver model (Section 2.4). Within EMDS, the basic products of a NetWeaver analysis are maps displaying the strength of evidence associated with the proposition for each logic topic (Figure 2). In the case of indicators evaluated in terms of multiple metrics (Table 2), map products assessing strength of evidence also were produced for each individual metric. The final ArcMap document (e.g., mxd file), including all map products of the TCA, is available in the supplementary materials section. After completing the full national TCA assessment for the continental U.S., results were parsed to each NFS Region and National Forest for subsequent use by these units. Within the overall scheme of the TCA process, it was envisioned that Regions and Forests could modify data inputs and NetWeaver logic as needed to improve the relevance of analytical products at the latter smaller spatial extents. Customizing the TCA for other spatial extents is addressed further in the Discussion.

3. Results

At a national scale, 55% of national forests and grasslands are in very good or good condition, whereas 26% are in poor or very poor condition (Table 3, Figure 3). Overall TCA condition ratings are based on simultaneous consideration of nine indicators and the twenty six metrics used to characterize indicators (Table 2). The importance of indicators varies geographically, and interpretations of conditions leading to an overall landscape ecosystem rating need to be made at a local scale. However broad generalizations can be made. The very poorest conditions are principally due to high insect and disease risk, extensive mortality occurring within the past 5 years, and high and very high wildfire hazard potential (Figures 4–6). Effects of elevated temperature and reduced precipitation, uncharacteristically severe or frequent wildfire, and fragmentation of habitat due to roads are also strongly associated with very poor and poor conditions (Figures 7–10).

Table 3. Frequency and areal distributions of the LTAs among the five overall ecological integrity ratings at the national scale.

Terrestrial Condition [1]	Frequency	Hectares	Percent of National Forest System Lands
Very Good	1618	15,862,119	18.41
Good	3962	31,896,498	37.02
Moderate	1736	15,942,480	18.50
Poor	1226	9,785,574	11.36
Very Poor	1491	12,669,501	14.71

[1] Classes used for classification of terrestrial condition in this table, and subsequent tables and figures, represent equal intervals on the NetWeaver scale for strength of evidence, with very good condition being ≥0.60, good condition being <0.60 and ≥0.20, etc.

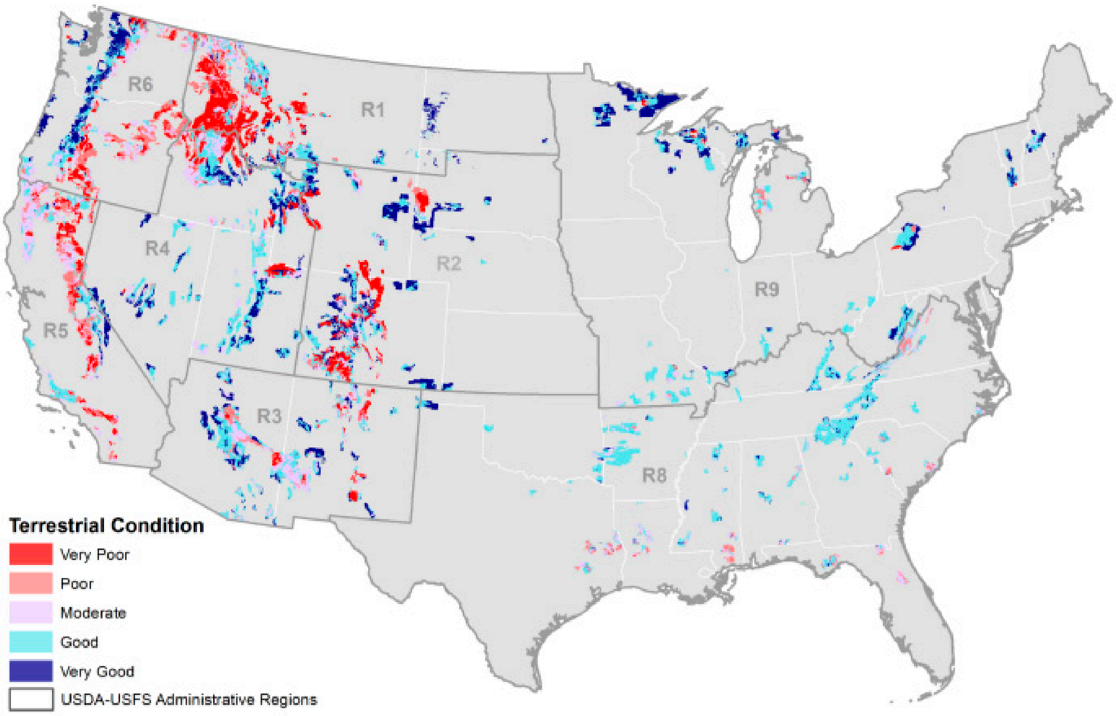


Figure 3. Overall ratings of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service administrative lands.

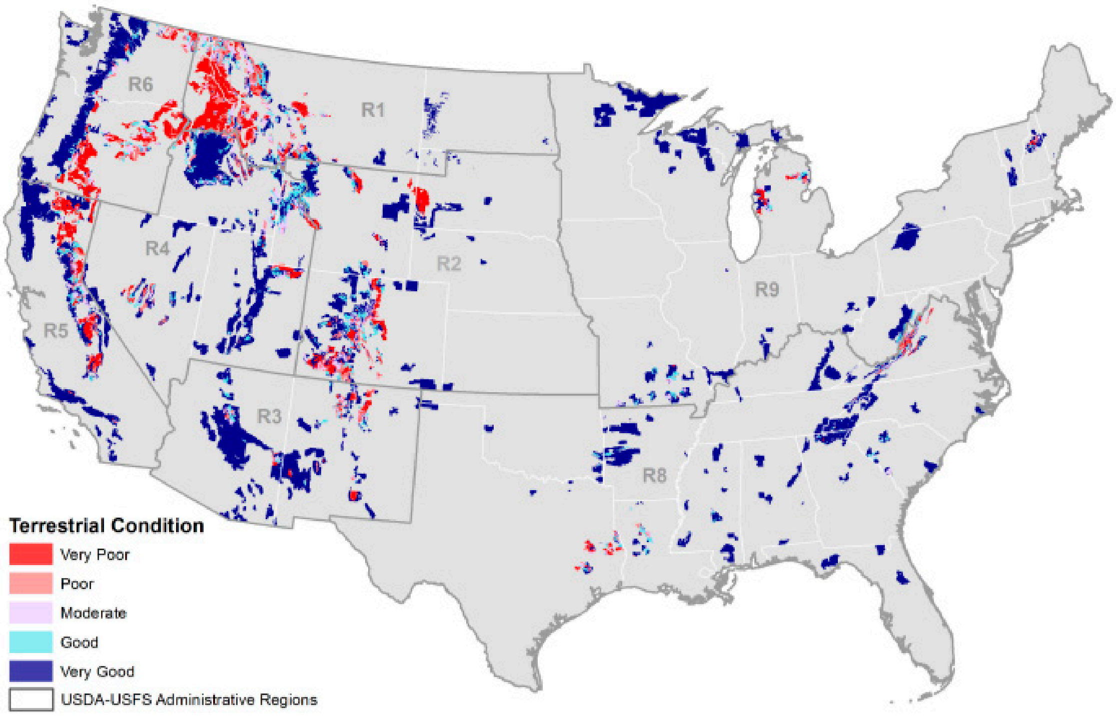


Figure 4. Ratings of the insect and disease risk metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

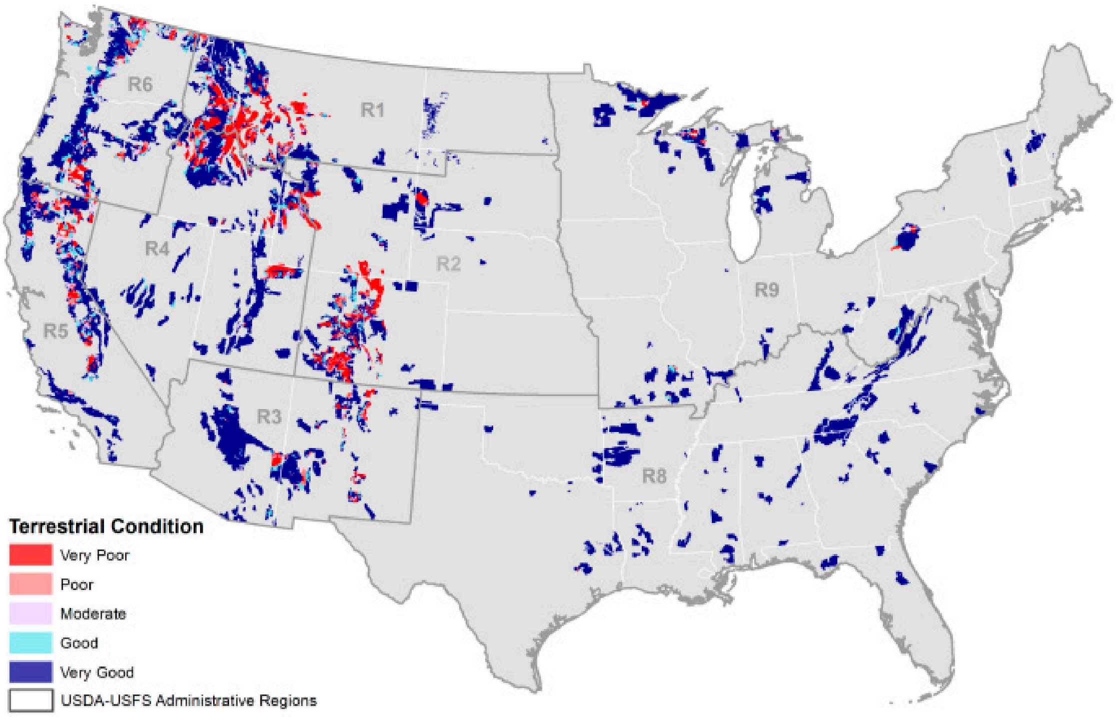


Figure 5. Ratings of the uncharacteristic tree mortality metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands. Tree mortality is based on 2010–2015 surveys of current mortality and excessive defoliation due to insect and disease outbreaks.

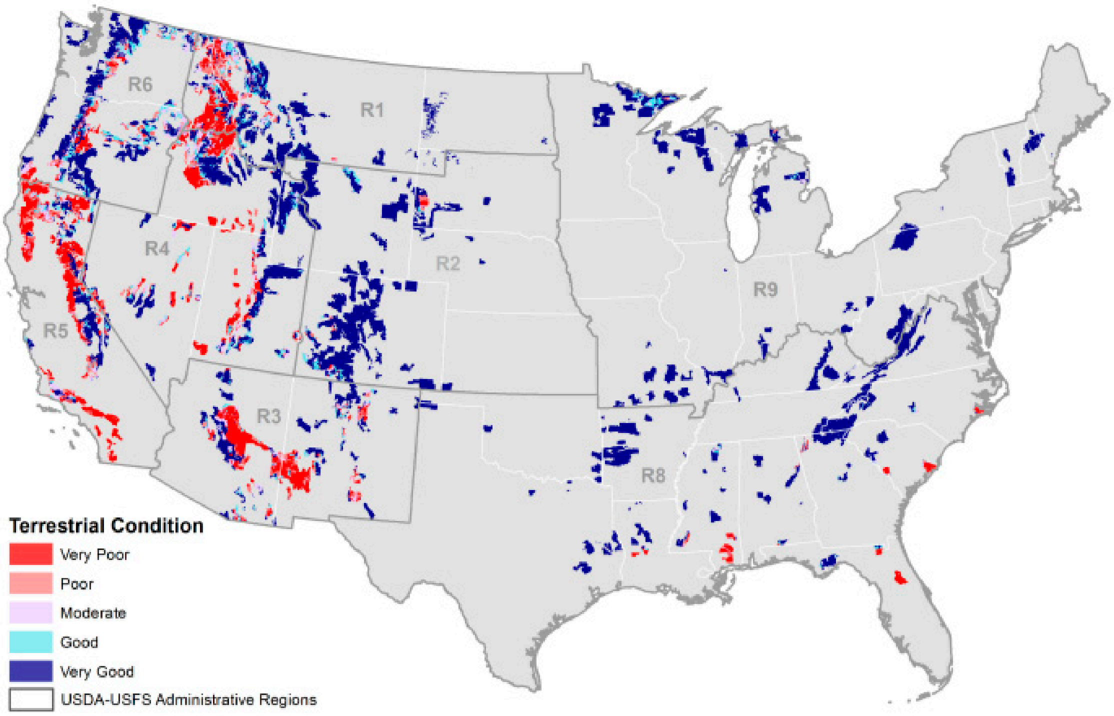


Figure 6. Ratings of the high and very high wildfire potential hazard metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

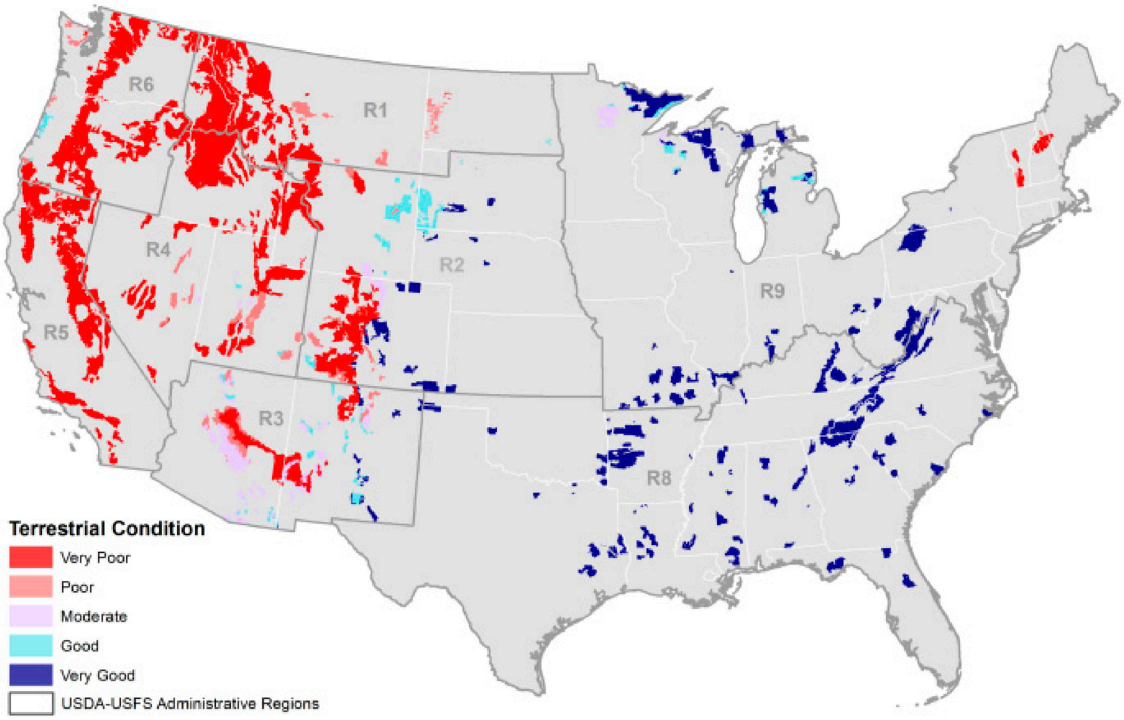


Figure 7. Ratings of the winter temperature shift metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

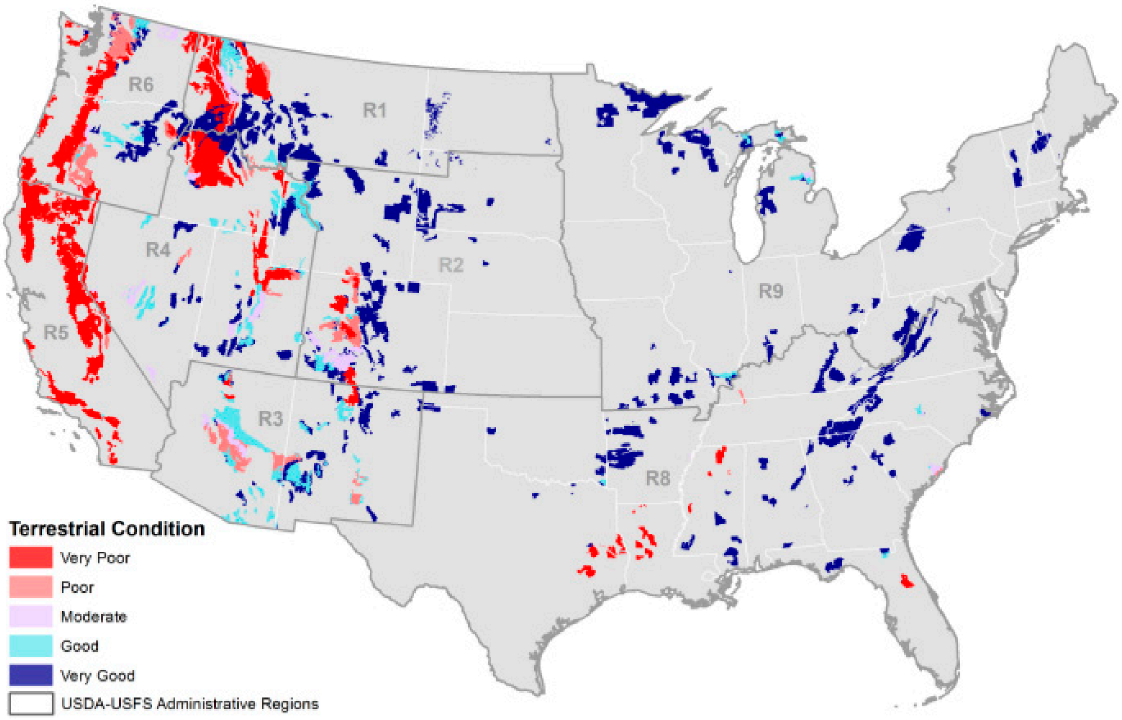


Figure 8. Ratings of the winter precipitation shift metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

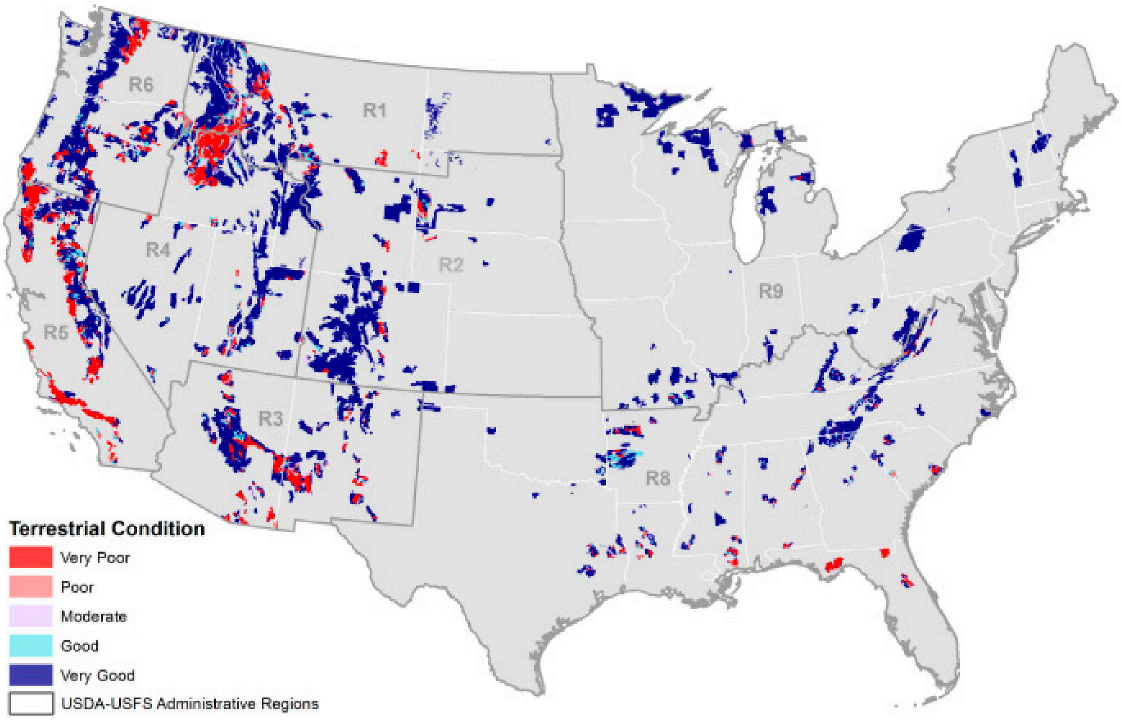


Figure 9. Ratings of the uncharacteristic wildfire indicator of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

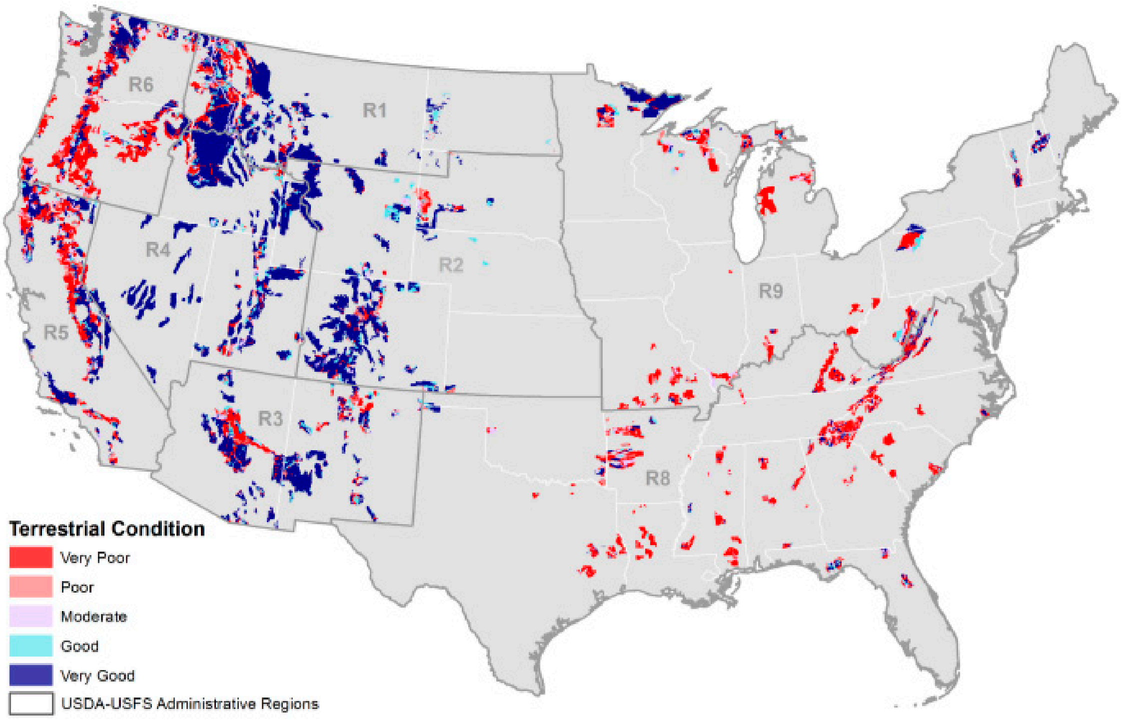


Figure 10. Ratings of the total road density metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

In general, conditions in the eastern United States are better than much of the west, largely because of low fire hazard potential and very limited extent of current mortality. The primary stressors affecting National Forests in the east include high road densities (Figure 10), air pollution (Figure 11), and vegetation departure from reference conditions (Figure 12). National Forests in the southeast are also being impacted by uncharacteristically severe or frequent fires and reduced spring and fall precipitation.

Poor and very poor conditions are concentrated in the western United States. Of the 64.8 million hectares occurring within western national forests' proclamation boundaries, 20 percent or 13.1 million hectares are at imminent risk of uncharacteristic mortality due to insects and disease, 9.4 percent or 6.3 million hectares have experienced mortality in the past five years, and 33 percent or 21.2 million hectares have high or very high wildfire hazard potential. Stressors of uncharacteristically severe wildfire and climate exposure (elevated temperatures, particularly winter temperatures, and reduced precipitation) are severe in the west but almost nonexistent in the east.

At a regional scale (see Figure 1 for boundaries of the USFS regions), Regions 1, 2, 5, and 6 have very large percentages of very poor and poor conditions (Table 4). Region 1 has extensive areas with high insect and disease risk (37% of the Region), high and very high wildfire potential (34% of the Region), and high mortality occurring within the past five years (16% of the Region). Region 2 has extensive areas with high insect and disease risk (20% of the Region) and high recent mortality (13.4% of the Region). Region 5 has extensive areas with high and very high wildlife potential (53% of the Region), high insect and disease risk (18% of the Region), high recent mortality (8.3% of the Region), and high road densities. Region 6 has high insect and disease risk (23% of the Region). All western Regions are experiencing stress due to elevated temperatures and to a lesser degree reduced precipitation. Of greatest concern are increases in winter temperature (Figure 7) and decreases in winter precipitation (Figure 8).

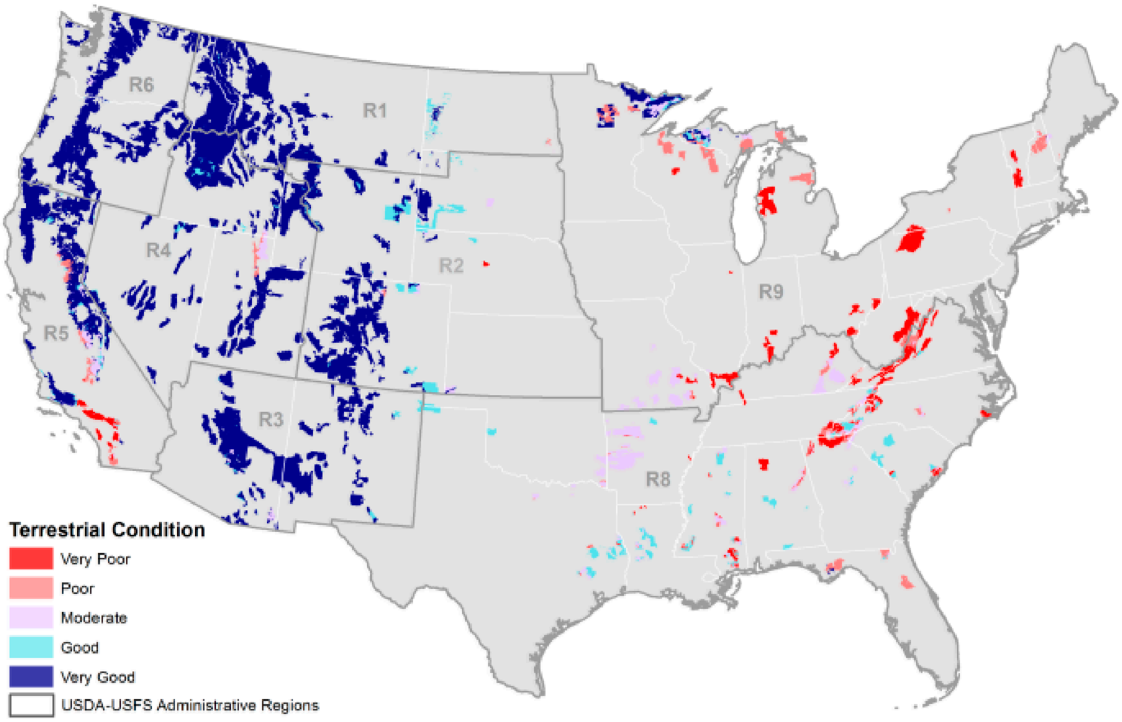


Figure 11. Ratings of the air pollution indicator of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

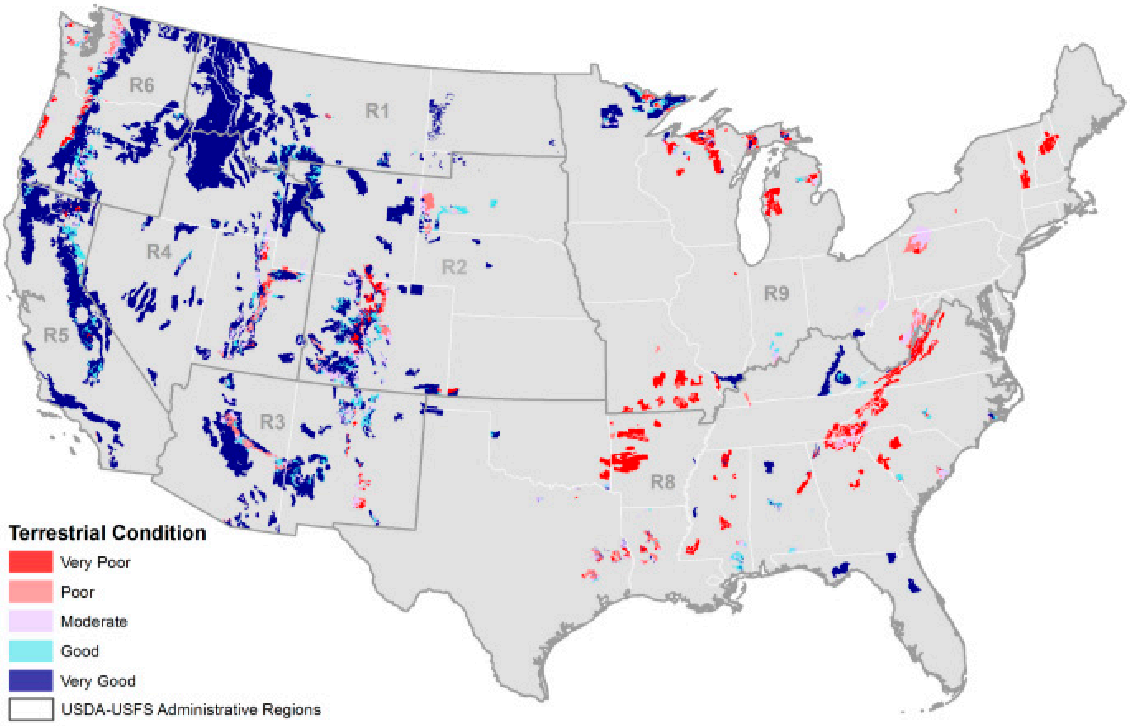


Figure 12. Ratings of the vegetation departure metric of the LTAs from the Terrestrial Condition Assessment on USDA Forest Service lands.

Table 4. Frequency and areal distributions of the LTAs among the five overall ecological integrity ratings at the regional scale.

Region	Terrestrial Condition	Hectares	Percent Regional	Region	Terrestrial Condition	Hectares	Percent Regional
1	Very Good	1,065,169	9.46	5	Very Good	448,620	4.74
1	Good	2,547,906	22.62	5	Good	1,894,951	20.01
1	Moderate	1,711,470	15.20	5	Moderate	3,263,593	34.46
1	Poor	1,497,393	13.30	5	Poor	2,162,259	22.83
1	Very Poor	4,439,859	39.42	5	Very Poor	1,702,486	17.97
2	Very Good	3,540,636	31.51	6	Very Good	2,295,182	20.36
2	Good	3,213,151	28.60	6	Good	2,573,660	22.83
2	Moderate	1,157,275	10.30	6	Moderate	2,514,413	22.30
2	Poor	874,659	7.79	6	Poor	2,337,587	20.73
2	Very Poor	2,449,247	21.80	6	Very Poor	1,554,411	13.79
3	Very Good	1,838,627	19.76	8	Very Good	550,467	5.36
3	Good	3,777,133	40.59	8	Good	6,794,328	66.13
3	Moderate	2,236,714	24.03	8	Moderate	1,860,729	18.11
3	Poor	834,357	8.97	8	Poor	1,060,174	10.32
3	Very Poor	619,630	6.66	8	Very Poor	8379	0.08
4	Very Good	2,805,318	20.08	9	Very Good	3,318,101	35.44
4	Good	6,056,478	43.35	9	Good	5,038,891	53.82
4	Moderate	2,741,347	19.62	9	Moderate	456,939	4.88
4	Poor	634,118	4.54	9	Poor	385,028	4.11
4	Very Poor	1,732,351	12.40	9	Very Poor	163,139	1.74

4. Discussion

Our results show a marked contrast between the eastern and western U.S., especially with respect to climate differences. The western United States has been subject to uncharacteristically severe wildfire in past decades largely due to a century of fire suppression, past logging, and climate exposure [28]. Fire suppression has resulted in increased tree densities and associated moisture demand, and increased fuel loads relative to historical or pre-European settlement forest conditions [29]. Increased winter temperatures reduce snowpack and water storage [30], and also reduce cold-induced mortality of damaging insects and diseases [31]. Increased temperatures during the growing season reduces fuel moisture, aggravating conditions promoting uncharacteristic wildfire [1,2], and increases the extent to which trees are stressed and less able to resist adverse effects of insect and disease.

Recent shifts in temperature affect western national forests far more than those in the east. The TCA used an increase of 1.11 °C (2 °F) as a threshold to identify LTAs undergoing recent severe temperature stress. Based on that threshold, 48% of NFS lands in the west are experiencing severe winter temperature stress in contrast to the less than one percent in the east. Spring, summer, and fall severe temperature stress affected 6.8%, 27.1%, and 24.8% of western national forests, respectively. Conversely, spring, summer, and fall severe temperature stress affected less than one percent of eastern national forests. Interactions leading to poor conditions, including altered landscape patterns, fuel complexes, incidence of insect and disease caused mortality, and climate-induced stress, are therefore manifest in the western United States far more so than the east.

We have presented the TCA framework used to complete a national level assessment of ecological integrity based on uncharacteristic stressors, conditions, and disturbance agents for national forest system lands in the United States. Results, data, and guidance on analytical procedures have been produced for agency applications, including a web map viewer and web-based information delivery system. Applications for performance accounting are being developed at a national scale. Regional applications that include use of the TCA in addition to regional data and assessments are being initiated. Local applications in land management planning are taking place on select national forests involved in the planning revision process. Moving beyond a national product to support regional and local applications of the TCA is one of the next phases of the project.

4.1. Customizing TCA for NFS Regions and National Forests

As mentioned in Section 2.5, the TCA assessment for the continental U.S. was intended as a starting point or template from which USFS Regions and National Forests could customize the assessment to make it more relevant to their local contexts. Customization of the analysis presented here can be done in at least four distinct ways:

1. National data presented in this study could be replaced with local data sources if local data sources were believed to be more accurate or more appropriate for the local context.
2. Thresholds used to define fuzzy membership functions that interpret the TCA metrics (Table 2) could be revised to better reflect local conditions. A good example in this context is the interpretation of road densities with respect to their effects on wildlife habitat fragmentation.
3. The national TCA logic includes several metrics related to uncharacteristic disturbances (indicator 6, Figure 2) including the spatial extent of mine impacts, landslides, blowdown, and flooding. Although logic topics and metrics associated with these impacts were designed into the NetWeaver logic model, they are turned off in the national analysis that we have presented because national data for these effects are not available. However, regions and forests could turn on one or more of these logic topics to include in their local assessments if they were considered important. Within the national TCA template, Region and Forest staff have two options for accounting for these ecosystem impacts: use of continuous measures (e.g., measured spatial extent) or use of ordinal rankings provided by specialists (see the HTML NetWeaver logic documentation included in the supplementary materials at the end of the paper).
4. Finally, the basic logic structure of the national TCA template is easily edited in NetWeaver by Region and Forest staffs to customize the logic for local contexts. For example, some logic topics in the national TCA template may not be considered relevant in some local contexts, in which case they can be turned off. In addition, the combination of logic operations used to synthesize evidence for logical premises of a particular logic topic might be edited by changing logic operators, or reorganizing the logic structure of premises to alter how a set of premises contribute to the strength of evidence for their parent topic.

The ability to customize the national TCA template for local application as described above creates some tension between assessments conducted at the different spatial extents of national, Region, and Forest. On the one hand, the national template was intended to promote, as far as practicable, consistency in how TCA assessments are conducted across spatial extents. On the other hand, an excessive emphasis on consistency across spatial extents has the potential to seriously compromise the utility of assessment products at more local extents. As a result, the NFS may need to consider an explicit governance process that balances the competing interests of national consistency and local relevance.

4.2. Additional Steps in Decision Support for Ecosystem Restoration and Maintenance

The results presented in this article evaluate the ecological integrity of LTAs on NFS lands in the continental US. However, in important respects, the analysis only represents the first step in a complete decision support process for the restoration of ecological integrity. In particular, the assessment characterizes terrestrial condition, which is an important foundation for a planning process, but it does not provide explicit support for implementing strategic and tactical planning decisions needed to meet restoration goals of the agency. As we discussed in Section 2.3, the EMDS framework includes a collection of decision engines that provide additional support for strategic and tactical decisions. In this section, we discuss how the associated decision support systems can be brought to bear to support management decisions for restoration and maintenance of LTAs.

The decision engine of CDP has been used for design of strategic multi-criteria decision models in EMDS since 2002 [15]. Whereas NetWeaver solutions describe the state of the system, strategic decision models assist resource managers with identifying which landscape units that are a high priority for management actions, by not only considering the state of the system, but by accounting for logistical considerations that are of practical importance to managers. Logistical considerations include such issues as feasibility, efficacy, cost, performance, consequences, social acceptability, etc., of potential management actions.

Whereas strategic decision models address the question of which landscape units are the highest priority for management, tactical decision models address the question of which management actions are the highest priority for any particular landscape unit, considering the biophysical context (or other contextual information) of the landscape unit. In other words, the strategic question concerns where, while the tactical question concerns what. Reynolds et al. [32] recently experimented with a CDP solution for tactical planning, however we believe that tactical decision models based on GeNIe and VisiRule may be more effective in tactical decisions, primarily because these systems can model more complex problems than CDP. For example, GeNIe supports sophisticated probabilistic reasoning based on Bayesian inference [33], and VisiRule, although providing a simple graphic interface, is supported by a powerful Prolog engine that allows very complex reasoning.

Reynolds et al. [32] also illustrated a variety of analytical sequences for decision support involving assessment and strategic and tactical planning, but more generally the architecture of EMDS was extensively re-engineered at version 5.0 to support the concept of workflows, by which any of the EMDS analytical components described above can be invoked in any sequence(s) (or series of sequences) needed to support spatial analysis and planning. EMDS currently supports Microsoft Windows Workflow for creating, running, and monitoring scientific workflows and Workflow NET. Interoperability of components is realized by data sharing, by which upstream analytical products are shared with any downstream analytical steps in an analysis sequence. In order to further extend interoperability in the workflow environment, EMDS now also implements Java script, R, and Python languages as tools for spatial data transformation.

5. Conclusions

The USDA Forest Service has recently completed an assessment of the ecological integrity of landscape ecosystems (LTAs) across its land base using the TCA framework. Results are beginning to be applied in national, regional, and local resource planning and management activities. Prospects for improving national performance accounting, and for developing strategic and tactical decision support systems that include social and economic considerations are being evaluated.

Supplementary Materials: The following are available online at 1. TCA metadata for data inputs to logic model https://www.cloudvault.usda.gov/index.php/s/5YHcvq213tFZQ3D; 2. Complete documentation of the NetWeaver logic in HTML https://www.cloudvault.usda.gov/index.php/s/rsHuooVYIk2xJLE; 3. ArcMap document with maps of all NetWeaver outputs for the TCA project https://www.cloudvault.usda.gov/index.php/s/11u5oCMxGvYAlhq.

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Author Contributions: All authors participated in the design and implementation of the TCA application; David Cleland and Robert Vaughan analyzed the data; Robert Vaughan did the geoprocessing to prepare data for EMDS; Keith Reynolds designed the NetWeaver logic and ran the analysis in EMDS; David Cleland, Barbara Schrader, and Larry Laing provided overall project oversight; David Cleland, Keith Reynolds, and Harbin Li wrote the paper.

Conflicts of Interest: The authors declare no conflict of interest.

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WINTER HABITAT AND NEST TREES USED BY NORTHERN FLYING SQUIRRELS IN SUBBOREAL FORESTS

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We determined characteristics of nest trees and surrounding habitats used by northern flying squirrels (*Glaucomys sabrinus*) in subboreal forests of northwestern British Columbia during winters 1996–1997 and 1997–1998. Nineteen radiocollared flying squirrels (12 males, 7 females) were located in 82 daytime nests. Animals used an average of 5.6 nest trees (±0.5 *SE*; range, 3–10) per animal. Core nest areas used by flying squirrels averaged 2.74 ± 0.62 ha in size; areas were more variable for males (range, 0.86–8.58 ha) than females (range, 0.03–2.23 ha). Nest trees were highly variable, suggesting that animals select more for suitable nest sites than for tree size: diameter at breast height was 16.7–79.0 cm, age was 42–174 years, and height was 11.2–32.7 m. A significant proportion of nest trees, however, were larger, older, and taller than trees that were randomly available in the locale of nest trees. Variation in habitats used by flying squirrels in the subboreal spruce (*Picea*) zone of British Columbia is evidence of the ability of this animal to occupy a wide range of conditions in a region that is not typified by old-growth forests.

Key words: *Glaucomys sabrinus*, nest-tree characteristics, radiotelemetry, winter habitat

The northern flying squirrel (*Glaucomys sabrinus*) occupies forested ecosystems across North America from Alaska and much of Canada to as far south as northern California in the west and North Carolina in the east (Wells-Gosling and Heaney 1984). Relatively few ecological studies have been conducted on this species because of its nocturnal and arboreal habits and small mass (about 150 g). As a cavity nester that is generally mycophagous, the northern flying squirrel has been considered a habitat specialist, dependent on old coniferous forests for shelter and food. In the northwestern part of its range, the species forages extensively on highly digestible mushrooms and supplements its diet with arboreal lichens when mushrooms are unavailable (Hall 1991; Laurance and Reynolds 1984; Maser et al. 1986; Zabel and Waters 1997). Consequently, northern flying squirrels potentially play a key role in maintenance of forest health by dispersing spores of mycorrhizal mushrooms (e.g., *Rhizopogon*) and fragments of arboreal lichens (e.g., *Bryoria*—Carey et al. 1999; Fogel and Trappe 1978; Hayward and Rosentreter 1994). Most recent attention has focused on their role as the main prey species of the endangered spotted owl (*Strix occidentalis caurina*) in the Pacific Northwest of the United States (Carey et al. 1997; Martin 1994).

Small changes in forest structure may have significant impact on habitat specialists. The extent of old-growth habitat specialization by northern flying squirrels in western coastal forests has been called into question recently by studies showing that the species is capable of subsisting in second-growth forests (Carey 1995; Martin

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1994; Rosenberg and Anthony 1992). Nonetheless, populations of northern flying squirrels have declined in the southeastern United States because of loss of forested habitat (Urban 1988). A similar trend has been observed for the ecologically similar eastern flying squirrel (*Pteromys volans*) in Finland (Hokkanen et al. 1982; Mönkkönen et al. 1997). In boreal ecosystems, very little is known about specific habitat requirements of northern flying squirrels. Only 2 northern studies have been reported: 1 in the boreal forests near Fairbanks, Alaska (Mowrey and Zasada 1984), and the other in mixed-wood forests of Alberta (McDonald 1995). There is a need for detailed research focusing on habitat requirements in areas not characterized by old-growth forests and during winter months, which are the most energetically stressful times of year for small mammals in northern regions.

To better understand habitat requirements during critical winter months, we investigated size of the core nest areas and characteristics of nest sites used by northern flying squirrels during winter in northwestern British Columbia. Our objectives were to determine the number of nest trees used per animal, identify structural attributes of nest trees and compare those features with randomly selected locations, quantify frequency of use of specific nest trees within the core nest areas by animals in winter, and examine distribution of nest trees among forest ecosystem types and seral stages of the study area.

MATERIALS AND METHODS

Study area.—Our study was conducted at 2 sites in northwestern British Columbia: the Smithers site, where most data collection occurred, and the Houston site, which was used to complement our sample size during the 1st field season when trapping success for flying squirrels was low at the Smithers site. The Smithers site was located in the Smithers Community Forest (54°43′N, 127°15′W), 10 km west of Smithers, British Columbia. The Houston site (54°27′N, 126°49′W) near Houston, British Columbia was about 26 km southeast of the Smithers site. Both sites are in the subboreal spruce biogeoclimatic zone (Pojar et al. 1987).

The Smithers Community Forest, about 4,620 ha, experienced fire disturbance in the 1930s and 1940s and has pockets of old-growth stands and mature trees scattered throughout younger stands. The study area was located on the lower slopes of Hudson Bay Mountain, with an average elevation of 850 m above mean sea level. Dominant species are hybrid white spruce (*Picea engelmannii* × *glauca*), subalpine fir (*Abies lasiocarpa*), lodgepole pine (*Pinus contorta*), and some trembling aspen (*Populus tremuloides*) and cottonwood (*Populus balsamifera*). The Houston site, at an average elevation of 585 m, has a similar disturbance history and plant species composition, with the addition of paper birch (*Betula papyrifera*). The site was surrounded by extensive clearcuts produced in the last 20 years. Witches' broom rusts (*Chrysomyxa*) were found on conifers at both sites.

Field methods.—Flying squirrels were captured using live traps (Tomahawk Live Trap, Model 201, Tomahawk, Wisconsin) in September and October 1996 and August and September 1997. Polyethylene stuffing was placed inside traps to provide thermal protection. Traps were covered with dark plastic garbage bags and mounted on trees about 1.5 m above and horizontal to the ground surface. Traps were baited with a mixture of peanut butter and rolled oats and were set at dusk and checked at dawn to minimize capture of nontarget species (e.g., red squirrels, *Tamiasciurus hudsonicus*, and martens, *Martes americana*). At the Houston site, we established 90 traps in 3 trapping grids with 50-m spacing. At the Smithers site, we set 156 traps along 7.5 km of an existing trail system, placing a trap on either side of the trail at about 50-m intervals.

Captured flying squirrels were transferred from the trap to a cloth and nylon-mesh handling cone. We anesthetized individual animals in a 4-l glass jar by wetting a gauze pad with isoflurane (Aerrane, Ohmeda Pharmaceutical Products, Mississauga, Ontario, Canada) and placing it in the bottom of the jar. An animal was kept in the handling cone for the sedation process so that it could be removed from the jar periodically to ensure adequate oxygen intake. Induction time was 5–45 min; recovery time was 5–20 min. Flying squirrels were weighed, sex and age were determined (using a combination of mass and pelage coloration to determine juvenile or adult age class—Davis 1963), and ear tags were applied (Monel No. 2, National Band and Tag Company, Newport, Kentucky). They were then fitted with temperature-sensitive radiocollars (Model PD-2CT, Hollohill Systems, Ltd., Woodlawn, Ontario, Canada) weighing about 3 g. We located flying squirrels in nest trees during the day using a Lotek receiver (Model SRX-400'A', Lotek Engineering, Inc., Newmarket, Ontario, Canada) equipped with a visual display of signal strength, which we used to distinguish the nest tree from other trees surrounding it. Animals were monitored 1–3 times weekly until mortality or loss of signal (2–6 months). Three animals located in a remote area of the Houston site were located only 3 or 4 times monthly after the 1st snowfall. The field season in 1996 was from September 1996 to March 1997; the field season in 1997 was from August 1997 to February 1998.

For each nest tree, we collected the following measurements: when and how often the site was used; tree species and, when possible, nest type (cavity, witches' broom, or a constructed nest, i.e., dray); tree height, measured with a clinometer, and nest height, if visible; tree diameter at breast height (dbh); tree age, using an increment borer; and Universal Transverse Mercator (UTM) coordinates, using a handheld base station–correctable global positioning system (GPS) unit (March II, Corvallis Microtechnology, Inc., Corvallis, Oregon). We differentially corrected UTM coordinates using the PC-GPS software (Version 2.50a, Corvallis Microtechnology). We measured canopy closure of overstory using both a concave spherical densiometer (Forest Densiometers, Bartlesville, Oklahoma) and a coverscope (Moosehorn CoverScope, Medford, Oregon). Four densiometer readings of overstory cover, taken about 1 m away from the nest tree facing cardinal directions, and 16 Moosehorn readings, taken at the same radius with 22.5° spacing between readings, were averaged for each nest site (Bunnell and Vales 1990; Cook et al. 1995); all readings were taken by the same observer. Wildlife tree classification, as defined by the British Columbia Ministry of Forests (British Columbia Ministry of Forests 1998; Guy and Manning 1994; Thomas 1979), was determined for each nest tree. That classification system rated 5 characteristics of the tree on a relative scale: visual appearance, crown condition, bark retention, wood condition (determined by examining the tree core extracted by the increment borer for decay), and lichen loading. The latter was estimated using the British Columbia Ministry of Forests Photographic Field Guide (Armleder et al. 1992), which rates abundance of lichens (*Bryoria* and *Alectoria*) on the lower 4.5 m of the tree, although we based our evaluation on a generalized overall rating for the entire tree. We used that guide to provide 4 relative classes of abundance: low (≤5 g of lichens/4.5 m of tree bole), moderate (5–50 g of lichens/4.5 m), high (50–250 g of lichens/4.5 m), and very high (250–625 g of lichens/4.5 m). We also determined a wildlife habitat value (high, medium, or low) for each nest tree, using a combination of species longevity, site position, decay value (based on the visual appearance rating for the wildlife tree classification), dbh, and tree height (Guy and Manning 1994).

Habitat characteristics around nest trees were measured during summer following each winter field season using nested plots of 5.6 and 10.6 m (Carey and Johnson 1995), with the nest tree at the center of each plot. Within the 10.6-m-radius plot, we recorded overall tree density (trees with dbh >7.5 cm), live-tree and snag densities, species composition of trees and dominant overstory species, abundance of arboreal lichens on each tree (using the same method as for nest trees), number of witches' brooms and visible cavities, and number of fallen trees (using 2 size classes of >7.5 cm dbh and <7.5 cm dbh). In the 5.6-m-radius plot, we measured density and species composition of saplings (>2 m tall, <7.5 cm dbh), understory cover (estimated visually in 3 classes: 0–10%, 10–50%, 50–100%), and dominant understory, midstory, and herb species. The biogeoclimatic ecosystem classification (Pojar et al. 1987) at each nest site was determined using the British Columbia Ministry of Forests Field Guide for the Prince Rupert Forest Region (Banner et al. 1993). That classification system was based on the soil moisture and nutrient regime, slope position, and vegetative species composition of the site. In addition, ecosystem mapping, which classified the area based on seral stage (related to stand structure), seral association (corresponding to successional status), and site units (describing climax potential), had been conducted at the Smithers site (MacKenzie and Banner 1991). The classification of each mapped unit (polygon) was based on differences in vegetative structure and composition and on landscape position. Seral stages were reported as shrub–herb, pole–sapling (10–30 years following disturbance), young–mature (young: 30–80 years; mature: 80+ years after stand disturbance), and old growth (150–250+ years old). Site descriptions of polygons were coded relative to gradients in soil moisture and nutrient regimes and have since been replaced by the above biogeoclimatic ecosystem classifications. To make both methods directly comparable, we determined an ecosystem type for each site description and biogeoclimatic ecosystem classification using 5 moisture and nutrient regimes (dry, mesic, mesic-wet, wet, and forested wetland).

We sampled 3 random sites for each nest tree. An initial bearing was randomly selected; the other 2 bearings were 90° and 180° from the first. A distance between 22 and 50 m was selected randomly for each bearing; 22 m was the required minimum to avoid overlapping of plots, and 50 m was set to limit sampling to an area in close proximity to the nest tree and readily accessible to an animal when selecting its nest site (Mowrey and Zasada 1984). At each random location, we designated the closest tree (with dbh >7.5 cm) as the random nest tree and the center of the nested plots for that sample. All measurements of tree and habitat characteristics were conducted as for nest trees, with the exception that Moosehorn coverscope readings were not taken at random nest trees.

Statistical analyses.—An α level of 0.05 was assumed for all analyses. Unless otherwise stated, all means are presented as ± *SE*. We used analysis of variance (ANOVA—Sokal and Rohlf 1995) to determine whether there were differences in number of nest trees used by flying squirrels between sites, years (to accommodate changes in habitat productivity), and sexes; sex was nested within either site or year. We limited our analysis to habitual nest trees, defined as trees in which an animal was located more than once. To determine if number of nests used by flying squirrels declined over time in response to increasing energetic demands of winter, we used a repeated measures ANOVA (Sokal and Rohlf 1995) to test for differences in the number of nests used per month among months and between years, sexes, and seasons (early winter, October–December; late winter, January–February). Only animals for which we had data spanning those 5 months were included ($n = 7$), and differences between sexes were examined within the same year (1997; 3 males, 2 females). We used correlation analyses (Moore and McCabe 1993) to examine relationships between number of nest trees used per animal and either number of times an animal was located or duration of time over which observations for that animal occurred. Animals located <10 times ($n = 4$) were excluded from analyses of number of nest trees, from calculations of the minimum and maximum distance between nest trees (computed using the PC-GPS software), and from calculations of the core nest area. Core nest areas, defined as the area enclosed by an individual's nest trees, were calculated using CALHOME (Kie et al. 1996). We used the 100% utilization distribution of the minimum convex polygon method (Jennrich and Turner 1969) because this method has the fewest assumptions related to how the area between nest trees was used by animals. Because data on core nest areas were not distributed normally and could not be transformed successfully, a Wald–Wolfowitz runs test (Siegel 1956), which includes an adjustment for small sample sizes, was used to test for differences between distributions of core nest areas of males and females. Levene's test for homogeneity of variances (Milliken and Johnson 1984) was used to test for differences in variance between core nest areas of males and females. We used correlation analyses to examine relationships between size of core nest areas and duration of time that animals were monitored or number of nest trees used per animal. Spatial distribution of nest trees used by aggregating animals and frequency of nest tree use were inspected visually.

To determine whether flying squirrels selected specific structural attributes for nesting, we divided several variables into classes to examine average percentage of observations per animal in each class. Nest trees were divided into 7 dbh classes (10-cm increments), 7 age classes (20-year intervals), and 5 height classes (10-m increments). We used Student's paired *t*-tests (Moore and McCabe 1993) to compare each structural attribute (dbh, age, height, and canopy closure) of each nest tree by animal to the average of its randomly sampled trees. We used 1-tailed analyses for all attributes except canopy closure because we hypothesized that flying squirrels select significantly larger, older, and taller trees for nesting. To determine frequency of occurrences in which each animal selected larger, older, or taller trees, we calculated the proportion of nest trees, by animal, that were larger than the average of associated random samples for each structural attribute. The average proportion across animals was compared (1-tailed Student's *t*-test) to a null hypothesis of 0.5, which would be expected if nest trees were selected at random, with the given attributes having no effect on nest-tree selection. Habitat characteristics around nest trees were compared with random locations using analyses similar to those for attributes of the nest tree: paired *t*-tests (2-tailed) and proportional differences. One animal was excluded from those analyses because it used only 2 nest trees, both of which were shared with another flying squirrel. A paired *t*-test compared densiometer and Moosehorn readings at each nest tree. We conducted Pearson chi-square (χ^2) contingency analyses (Everitt 1977) of frequency data to determine whether species composition of nest trees deviated from randomly sampled trees by site and whether the biogeoclimatic ecosystem classification differed between nest trees and random sites or between classifications determined on site at nest trees and those obtained from polygon descriptions on the ecosystem map of the Smithers Community Forest. Descriptive statistics, *t*-tests, correlations, tests of normality, nonparametric tests, and all graphical representations were completed using STATISTICA (StatSoft, Inc. 1997).

RESULTS

Nest use.—Nineteen northern flying squirrels (12 males, 7 females) were radiocollared and monitored over the 2 field seasons. We located animals 568 times in 82 daytime nest trees. Squirrels used an average of 5.6 ± 0.5 nest trees/animal (range, 3–10 trees). When occasional nest trees, defined as trees in which an animal was located only once, were removed from the data set, average number of nest trees used habitually per animal was 3.8 ± 0.4. There were no differences in number of habitual nest trees per animal between sites, years, or sexes (all $P > 0.182$). As determined by repeated measures ANOVA, number of nest trees used per month did not differ among months or between years or sexes (Fig. 1). There was a trend for the number of trees used per month in early winter to be higher than in late winter for the 7 animals that were alive throughout the 5-month winter period ($F = 4.89$, *d.f.* = 1, 6, $P = 0.069$).

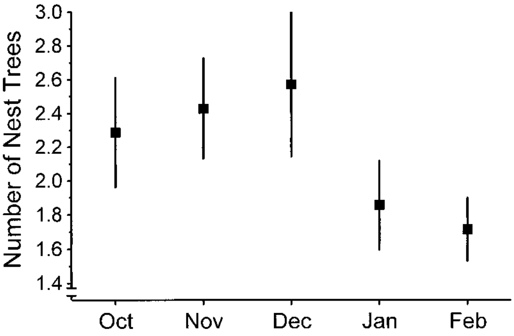

FIG. 1.—Average number of nest trees (± *SE*) used monthly during early winter (October–December) and late winter (January–February) by 7 northern flying squirrels in northwestern British Columbia.

On 280 occasions, individual flying squirrels were relocated on consecutive days. For 92% of those observations, animals stayed in the same nest tree the 2nd day. For the remaining observations, where animals moved to a new nest tree, average distance moved was 163.2 ± 21.9 m but ranged from 7.5 to 362.7 m. There was no correlation between number of nest trees located per animal and number of observations per animal ($r = 0.55$, $P = 0.058$) or time span over which observations occurred ($r = 0.26$, $P = 0.394$).

Core nest areas.—Core nest areas used by flying squirrels averaged 2.74 ± 0.62 ha. Distribution of sizes of core nest areas differed between males and females (adjusted $Z = 2.072$, $P = 0.038$). Males used a wider range of sizes (0.86–8.58 ha) than did females (0.03–2.23 ha) and had a higher variance ($F = 11.181$, *d.f.* = 1, 13, $P = 0.005$). Size of core nest areas was not correlated with length of time an animal was monitored ($r = 0.28$, $P = 0.320$, $n = 15$) but was correlated positively with number of nest trees used by the animal (r = 0.58, P = 0.022, n = 15). When 2 males whose core nest areas contained large sections that were not used by the animals (1 animal moved to a new area and the other core nest area contained a road) were removed from the analysis, the relationship was even stronger (r = 0.75, P = 0.003, n = 13). The smallest distance between nest trees in the core nest area for each animal averaged 60.1 ± 15.5 m (range, 7.5–203.3 m); maximum distance between nest trees averaged 361.2 ± 42.7 m (range, 78.4–751.4 m). Average maximum distance between nest trees was larger for males (435.7 ± 51.1 m) than for females (249.5 ± 48.0 m; t = 2.51, *d.f.* = 13, P = 0.026).

Spatial and temporal use of nest trees varied among individual animals. Some used predominantly 1 or 2 nest trees in their core nest areas (Fig. 2A); others used several trees relatively uniformly throughout the field season (Fig. 2B). Use of individual nest trees ranged from 1.2% to 85.5% of the total number of locations for an animal in its core nest area. Overlap in core nest areas occurred when 2 radiocollared animals used the same nest tree but at different times. That situation occurred twice, once with a tree used by animals in different years and once during the same winter season. Overlap in core nest areas also occurred in the case of aggregating animals (Cotton 1999). In those instances, 2 or 3 radiocollared animals shared 3 or 4 nest trees for 1–2 months.

Habitat characteristics.—Characteristics of nest trees were variable: dbh ranged from 16.7 to 79.0 cm (33.3 cm ± 13.3 *SD*), age from 42 to 174 years (83.2 ± 22.7 years), and tree height from 11.2 to 32.7 m (22.2 ± 4.7 m). Most nest trees used by each animal were 25–35 cm dbh, 60–80 years old, and 20–25 m tall (Fig. 3). Of the 18 animals for which nest tree characteristics were compared with random samples using paired t-tests, only 4 animals selected trees with significantly larger dbh trees, 4 selected taller trees, and 3 selected older trees than the associated random nest trees. However, an inherent problem with the paired t-tests (by animal) was that the magnitude of 1 comparison may have a strong effect on the other comparisons in the set if there is high variation among values. In the case in which an animal chose a very small nest tree, the large difference between that tree and random nest trees would overwhelm paired t differences for the animal's other trees, even if those nest trees were larger than associated random nest trees. Therefore, we examined proportions of nest trees that were larger, older, or taller to weight each nesting choice equally. When analyzed relative to frequency of selecting those attributes, a significant proportion of nest

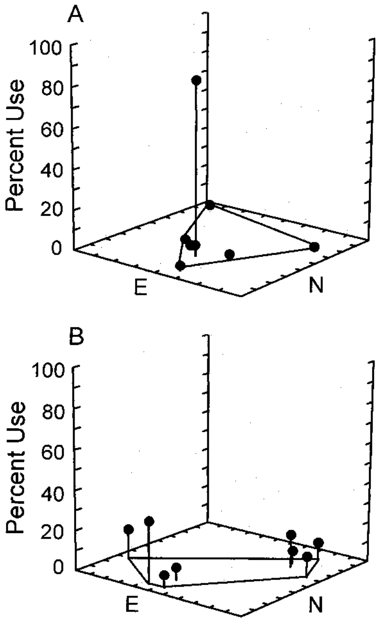

FIG. 2.—Examples of the spatial distribution (UTM coordinates in the x–y plane) and percentage of use of individual nest trees by northern flying squirrels (as a percentage of the total number of observations for the animal within a field season; z-axis). Solid symbols represent individual trees; each grid cell in the x–y plane = 0.25 ha. A) Male at the Houston site, September–March, 8 nest trees. Outlined core nest area = 5.09 ha. B) Male at the Smithers site, September–February, 8 nest trees. Outlined core nest area = 8.58 ha.

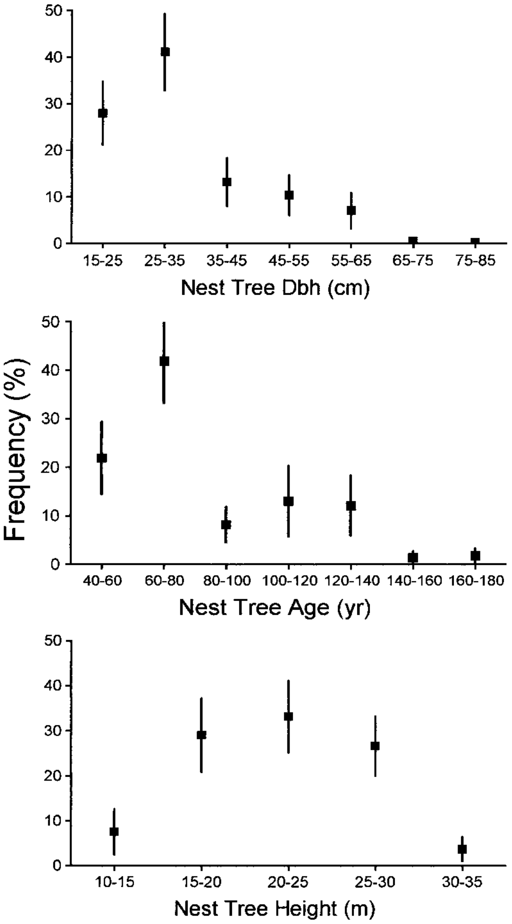

FIG. 3.—Tree characteristics ($\bar{X} \pm SE$) of nest trees ($n = 82$) used by 15 northern flying squirrels in northwestern British Columbia.

TABLE 1.—The mean proportion (averaged across animals, $n = 18$) of nest trees used by northern flying squirrels in northwestern British Columbia that were larger than the average of the associated random samples for diameter at breast height (dbh), age, and height (tested against a null hypothesis of 0.5).

Characteristic	Proportion $\bar{X}$	Proportion *SD*	t_{17}	*P*
dbh (cm)	0.771	0.188	6.116	<0.001
Age (year)	0.657	0.273	2.436	0.013
Height (m)	0.756	0.211	5.148	<0.001

trees used by flying squirrels were larger in dbh, age, and height than the average of the associated random samples for each tree (Table 1). Canopy closure was greater at nest trees when recorded with a densiometer (77.4% ± 1.8%; range, 24.2–98.7%) than when measured with a Moosehorn coverscope (72.2% ± 2.4%; range, 27.5–100%; $t = 2.74$, *d.f.* $= 81$, $P = 0.007$). Canopy closure at nest sites did not differ from that at random sites.

The wildlife tree classification indicated that 91.5% of nest trees had intact crowns, 85.4% of trees had minimal (<5%) bark missing, and 70.7% of trees had relatively sound wood with limited or essentially no decay present, as determined from core samples taken at a height of 1.3 m. Abundance of arboreal lichens (*Bryoria*, *Alectoria sarmentosa*) on nest trees was low to moderate (≤50 g of lichens/4.5 m of tree bole) for 92.7% of selected trees and was similar within nest-tree habitat plots and at random sites. Wildlife habitat value, as defined by Guy and Manning (1994), was high for 6.1% of nest trees, medium for 87.8% of nest trees, and low for 6.1% of nest trees.

Species composition of nest trees (Table 2) differed from the randomly sampled trees at the Smithers site (Pearson $\chi^2 = 12.741$, *d.f.* $= 2$, $P < 0.002$) but not at the Houston site (Pearson $\chi^2 = 1.869$, *d.f.* $= 1$, $P = 0.172$). Only 3 nest trees were snags (1 hybrid white spruce, 1 lodgepole pine, 1 aspen), and only 14 had visible nests (11 witches' brooms and 2 drays on hybrid white spruce, 1 cavity in a lodgepole pine). Those nests were at an average height of 11.5 ± 1.1 m. All other nesting sites were presumed to be in cavities or nest structures that were not visible from the ground.

Habitat characteristics in plots surrounding nest trees also were variable (Table 3). When compared with associated random plots using paired *t*-tests, only 1 animal used nest trees with significantly greater surrounding tree density, 1 animal used nest trees with significantly greater surrounding density of snags, and another had a lower

TABLE 2.—Species composition (%) of nest trees used by northern flying squirrels and randomly sampled trees at 2 study sites in northwestern British Columbia.

Site	*n*	Hybrid white spruce	Lodgepole pine	Subalpine fir	Aspen	Cottonwood	Birch
Smithers							
Nest trees	52	59.6	26.9	7.7	3.8	1.9	0
Random trees	156	40.5	25.9	32.9	0.6	0	0
Houston							
Nest trees	30	66.7	33.3	0	0	0	0
Random trees	90	73.9	19.6	5.4	0	0	1.1

density of snags surrounding its nest trees. Two flying squirrels used nest trees with fewer large fallen trees (>7.5 cm dbh) surrounding them, whereas 1 animal had fewer small fallen trees in the surrounding plots than in associated random samples. Average proportion of plots in which tree density surrounding nests were greater than that of associated random plots did not differ from the null hypothesis nor did average proportion of plots that had lower densities than those found in associated random plots. The number of witches' brooms per 353-m^2 plot surrounding nest trees ranged from 0 to 11. Dominant overstory species generally were hybrid white spruce or lodgepole pine, with subalpine fir or hybrid white spruce as the dominant regenerating midstory species. Dominant understory species included black huckleberry (*Vaccinium membranaceum*), thimbleberry (*Rubus parviflorus*), purple peavine (*Lathyrus nevadensis*), and red-stemmed feathermoss (*Pleurozium schreberi*), which are some of the common indicator species in the biogeoclimatic ecosystem classification. Understory cover of herbs and nonwoody shrubs was high (>50% cover) for the majority (65.9%) of nest tree plots.

TABLE 3.—Habitat characteristics (per 100 m^2) surrounding 82 nest trees used by 19 northern flying squirrels in northwestern British Columbia.

Habitat characteristic	$\bar{X}$	*SD*	Range
Overall tree density	12.3	6.6	1.7–34.3
Live tree density	10.4	5.8	1.4–32.6
Snag density	1.9	1.6	0.0–7.1
Small fallen trees (<7.5 cm diameter)	6.5	7.7	0.6–44.3
Large fallen trees (>7.5 cm diameter)	4.4	2.5	0.3–14.6
Sapling density	10.0	14.5	0.0–77.0

Ecosystem types around nest trees did not differ significantly from random locations at either study site (Table 4). The biogeoclimatic ecosystem classification of nest trees was the same as the associated random samples 64% of the time. Mesic and mesic-wet types were most common, with 11 of 18 animals using >1 type of ecosystem. Ecosystem types, as determined from the ecosystem mapping of the Smithers Community Forest, did not always match on-site determinations (19 of 52 comparisons), but they did not differ significantly ($\chi^2 = 6.356$, *d.f.* = 3, $P = 0.096$; Table 4). Map polygons usually were classified as the next most closely related ecosystem type when there was a discrepancy (16 of 19 comparisons). Distribution of nest trees at the Smithers site by seral stage of the stands (also determined from the ecosystem maps) was 1.9% in shrub–herb stands, 38.5% in pole–sapling stands, 53.8% in young–mature stands, and 5.8% in old-growth stands. However, 45% of the nest locations in pole–sapling and 7.1% of the locations in young–mature seral stages occurred in 4 polygons that also contained mature trees remaining in the stand after disturbance. Pole–sapling and young–mature stands were the most

TABLE 4.—Comparisons of ecosystem types around nest trees used by northern flying squirrels and their associated random samples and of classifications determined for nest trees on site and from ecosystem maps (MacKenzie and Banner 1991) at the Smithers site.

		Occurrence of ecosystem type (%)				
	n	Dry	Mesic	Mesic-wet	Wet	Forested wetland
Ecosystem type						
Nest trees	82	11.0	39.0	30.5	18.3	1.2
Random locations	246	10.8	39.2	38.0	11.2	0.8
Nest-tree classification						
On site	52	1.9	42.3	25.0	28.8	1.9
Ecosystem maps	52	5.8	40.4	38.5	11.5	3.8

common stand types in the area, and only a few pockets of old-growth stands were available to animals (Cotton 1999). At the Smithers site, 8 of the 12 animals used nest trees in 2 types of seral stages as determined from the ecosystem maps; the rest of the animals used only 1 type.

DISCUSSION

Core nest areas.—Northern flying squirrels occupied core nest areas that were variable in size and used a variable number of nest trees. We defined core nest areas for flying squirrels instead of home ranges because our data reflect only nest sites and may not incorporate all foraging areas. Nighttime telemetry efforts to delineate foraging areas were not successful (Cotton 1999). Thus, our values of core nest area may be smaller than those reported in studies of traditional home ranges. Home ranges were similar in Oregon (Martin and Anthony 1999; Witt 1992) and West Virginia (Urban 1988), although sample sizes were low in most areas (Table 5). Unfortunately, effects of small sample size and different

TABLE 5.—Size of home ranges reported for northern flying squirrels in North America.

Location	Home range (ha)[a]	Range (ha)	Sex	*n*	Method	Source
Northwestern British Columbia	3.7 ± 0.9	0.9–8.6	Males	9	Minimum convex polygon[b]	Current study
	1.4 ± 0.4	0.03–2.2	Females	6		
Northwestern British Columbia	10.3[c]	2.1–14.5	Sexes combined	5	Minimum convex polygon[b]	T. Mahon and D. Steventon (in litt.)
Western Oregon	4.2 ± 0.3	3.4–4.9	Sexes combined	4	Minimum convex polygon	Witt (1992)
Central Oregon	5.9 ± 0.8	2.6–17.0	Males	20	Adaptive kernel	Martin and Anthony (1999)
	3.9 ± 0.4	1.9–8.0	Females	19		
New Brunswick	12.5[d]	2.7–17.0	Males	7	Minimum convex polygon	Gerrow (1996)
	2.8[d]	2.2–6.9	Females	8		
West Virginia	5.2 ± 1.1	3.1–6.8	Males	3	Modified minimum area	Urban (1988)

[a] Mean ± *SE*.

[b] Calculated from nest-tree locations only, without observations of animal activity.

[c] Mean only.

[d] Median only.

sampling techniques limit direct comparisons of home-range size among geographic areas and forest types. We observed that sizes of core nest areas used by males were more variable than those of females. Males that occupy larger territories may have greater access to females than do males with smaller areas (Gerrow 1996; Martin and Anthony 1999). Males also showed a larger average maximum distance between nest trees than did females. Other studies support a difference in home-range sizes between males and females for northern flying squirrels (Gerrow 1996; Martin and Anthony 1999) and for closely related southern flying squirrels (Bendel and Gates 1987).

Use of multiple nests by northern flying squirrels has been suggested to be an adaptive response to variable food abundance (Carey et al. 1997). For example, in early winter, animals likely forage on widely dispersed mushrooms, whereas in late winter, they may rely on more readily available food sources, such as arboreal lichens or cached fungi. It is not known if flying squirrels cache fungi, but Mowrey and Zasada (1984) frequently observed flying squirrels stealing cached fungi from middens of red squirrels. Molds were found in diet samples from gastrointestinal tracts in winter and in fecal pellets in spring and summer from northern flying squirrels of the boreal mixed-wood forests of Alberta, suggesting that the food had been cached prior to consumption (R. S. Currah, pers. comm.). Average number of nests (5.6) used by animals in our study was similar to that reported in coastal forests of western Oregon ($\bar{X}$ = 6.1—Carey et al. 1997), but less than noted in interior forests of Alaska ($\bar{X} > 8$—Mowrey and Zasada 1984). Average distance moved between consecutive daily locations (163 m) was larger in our study than in central Oregon (71 m—Martin and Anthony 1999) but was similar to the distance between consecutive nest trees reported for coastal forests of Oregon (Carey et al. 1997).

Spatial and temporal use of nest trees did not follow a consistent seasonal pattern, leading to use of more nest trees than might be expected if food became very patchily distributed in winter or use of fewer nest trees as energetic demands increased with winter severity. Instead, animals had individual strategies. Some used many of their nest trees throughout winter; others were extremely faithful to only 1 or 2 nests (Fig. 2). We observed both strategies in both field seasons. Those strategies may be influenced by social factors that we were unable to measure, such as competition for nest sites by conspecifics and other species (e.g., red squirrels) and occurrence of aggregations with other flying squirrels. We observed overlap of core nest areas during aggregation and also when nest trees were used by >1 animal without aggregation. Overlap of core nest areas suggests overlap of home ranges. This overlap is not unusual given the diversity of tree characteristics and ecosystem types in the area and the social nature of the animals, as seen by aggregating behavior (Carey et al. 1997; Mowrey and Zasada 1984). Gerrow (1996) reported that males and females often foraged together in New Brunswick, Canada; females showed very little overlap of home ranges, whereas home ranges of males often overlapped and encompassed large parts of smaller female home ranges.

Habitat characteristics.—Northern flying squirrels showed considerable flexibility in the characteristics of the nest trees that they selected. Size (dbh and height) and age of nest trees were extremely variable, ranging from 50% to 150% of mean values. None of the animals in our study nested only in the largest nest trees; rather, animals used 3–10 different and highly variable trees. Animals did not have access to high numbers of large, old trees in the study area; in comparison to trees that were randomly available in the locale of nest trees, however, animals selected a significant proportion of trees that were larger, older, and taller. Martin (1994), Gerrow (1996), and Carey et al. (1997) also showed that flying squirrels selected larger nest trees when available. Our data provide further evidence that flying squirrels are not limited to old-growth habitats, but within younger stands, they select the largest trees available.

Mean values of characteristics of nest trees determined in our study are comparable to findings in interior Alaska (Mowrey and Zasada 1984), central British Columbia (Peterson and Gauthier 1985), Alberta (McDonald 1995), and 2nd-growth forests of central Oregon (Martin 1994; Table 6). Researchers in western Oregon reported larger nest trees (Carey et al. 1997; Witt 1992). Given the combination of tree species and coastal climate, forests of western Oregon typically are characterized by larger trees than are interior forests. The larger trees used by flying squirrels likely reflect this increased availability of large trees. In contrast, nest trees used by flying squirrels in New Brunswick were shorter than those in other studies. In all studies, however, a wide range of dbh and height was reported for nest trees. Given this variation, it appears likely that northern flying squirrels select for trees with suitable nest sites rather than for tree size, but suitable nest sites are more likely to occur in larger, older trees, which are more prone to internal decay (Lewis and Lindgren 1999).

Canopy closure around nest trees was variable. We used 2 methods to measure canopy closure because of recent studies indicating that spherical densiometers are biased towards overestimating cover (Bunnell and Vales 1990; Cook et al. 1995). Our results support those findings. The Moosehorn coverscope had a limited, more variable projection of overstory cover, whereas we observed smaller standard deviations and consistently larger readings at each nest tree using the spherical densiometer.

The high percentage (96.5%) of live trees used as nest trees differed from the common view of a ‘‘wildlife tree’’ as a decaying snag. The low percentage of nest trees classified as having high wildlife habitat value (6.1%) occurred because the majority of nest trees were live trees that were of smaller diameter (<50 cm) and height (<20 m) than the highest rated class (Guy and Manning 1994). The appropriateness of the variables used in this classification system for determining habitat value for northern flying squirrels seems questionable. Abundance of lichens on nest trees was not different from that in random samples, but arboreal lichens (*Bryoria*) were present at every nest site and throughout the stand. Hence, nest-site selection by flying squirrels in our study probably was not limited by availability of arboreal lichens, which are consumed and also used as nesting materials (Hayward and Rosentreter 1994; Maser et al. 1985).

The most common species of trees used for nesting by northern flying squirrels in northwestern British Columbia were hybrid white spruce and lodgepole pine. The main difference in species composition at the Smithers site between nest trees and random samples was the low use of subalpine fir. Subalpine fir tends to decay faster than other conifers in the area, making it a likely species for natural and excavated cavities. Species composition and accompanying seral stages within the study area, however, have been determined largely by the fire disturbance ecology. Spruce and pine are the dominant overstory species, with subalpine fir naturally regenerating as the dominant midstory species. Much of the subalpine fir in the area is not as old or large as the spruce and pine. Consequently, subalpine fir could become more important as a nest-tree species as the stands mature.

Use of snags as nest trees was relatively low (3.5%) in our study, similar to findings in Oregon (Carey et al. 1997; Martin 1994). It has been suggested that live trees may be more suitable as nest sites for cavity nesters because overhead branches provide protection from weather, increased cover, and structural complexity for predator avoidance and escape and because of the longer persistence of live trees as compared with

TABLE 6.—Diameter at breast height (dbh) and height of nest trees used by northern flying squirrels in North America.

Location of study	Tree type	*n*	dbh (cm) $\bar{X}$	dbh (cm) Range	Height (m) $\bar{X}$	Height (m) Range	Source
Northwestern British Columbia	Conifer	79	31.5	16.7–79.0	19.3	11.2–32.7	Current study
	Deciduous	3	33.4	19.9–40.5	22.3	13.9–26.4	
Northwestern British Columbia	Conifer	15	33.7				T. Mahon and D. Steventon (in litt.)
Interior Alaska	White spruce	32	32.6	10.4–56.1	24.1	8.5–38.4	Mowrey and Zasada (1984)
	Paper birch	5	21.9	18.0-28.2	12.6	5.8–17.4	
	Trembling aspen	6	30.6	27.2–35.1	15.6	13.4–21.3	
Central British Columbia	Nest trees with cavities	6	30.4				Peterson and Gauthier (1985)
Central Oregon	Second growth	65	36.0				Martin (1994)
	Old growth	43	101.0				
Western Oregon	Conifer	7	66.1	16.0–88.0	19.9	19.0–40.0	Witt (1992)
Western Oregon	Second growth, with thinning						Carey et al. (1997)
	Live	118	60.1		37.6		
	Snags	67	41.6		11.0		
	Second growth, with mature trees						
	Live	186	49.0		32.6		
	Snags	86	63.7		12.6		
Alberta	Deciduous		36.5		16.6		McDonald (1995)
New Brunswick	Conifer, with broom or dray	55	28.6	7.0–57.0	14.5	4.0–22.0	Gerrow (1996)
	Conifer and deciduous						
	Woodpecker cavities	33	24.5	11.5–47.0	7.4	1.6–24.0	
	Natural cavities	42	29.7	15.0–69.0	9.8	2.0–21.0	

snags (Carey et al. 1997). In contrast, McDonald (1995) reported that 59% of the nest trees used by flying squirrels in the mixed-wood forests of Alberta were snags. Gerrow (1996) also found that when cavities were used in New Brunswick, trees were often snags, but nest use was linked closely to availability; cavities were used where abundant and witches' brooms were inhabited where they were readily available. The low use of snags and brooms as nest sites (13.4%) in our study likely reflects the relatively young age of the stands. Mowrey and Zasada (1984) found northern flying squirrels primarily in witches' brooms in Alaska and stressed the importance of brooms for aggregations of animals. In our study, brooms were used in only 2 of the 9 nest trees in which we observed aggregations of radiocollared animals (Cotton 1999). It is unknown if animals in other brooms were nesting with flying squirrels that were not radiocollared. We elected not to climb nest trees to investigate because Carey et al. (1997) reported that during 10 of 12 climbs to determine nest type, flying squirrels subsequently left the tree and did not return.

Nest trees used by flying squirrels were located in areas with a high degree of tree regeneration (>1,000 saplings/ha—Table 3) and numerous fallen trees that provided substantial amounts of coarse woody debris. Flying squirrels, however, did not appear to select particular habitat characteristics at nest sites that differed from random sites. Most other researchers have reported similar results (Gerrow 1996; Martin 1994; Payne et al. 1989; Rosenberg 1990; Urban 1988), although presence of large snags (>50 cm dbh) in coastal Oregon was important, and flying squirrels in central Oregon avoided areas with high densities of small snags (Carey et al. 1997; Martin 1994). In southwestern Oregon, habitats used by flying squirrels had a high degree of decadence (including snags and logs) and complex canopies (Carey et al. 1999). We suggest that the wide range in habitat attributes observed in our study is further evidence of the flexibility of these animals and an indication of structural diversity within the stand.

Nest trees tended to be in mesic and mesic-wet areas. Those sites were rich in soil moisture and nutrients and exhibited high species diversity and structural complexity in shrub and herb layers. Such sites likely produce more mushrooms, a key component in the diet of flying squirrels (Waters and Zabel 1995). Distance between nest sites and random sites in our study may not have been great enough to reflect true availability of all ecosystem types at the landscape level. Large polygons, however, often contain pockets of other ecosystem types. We are confident that we could detect presence of these pockets and did not observe flying squirrels selecting 1 particular type of ecosystem. Most nest-tree locations (>92%) were in pole–sapling or young–mature seral stages, but 21% of the nest trees at the Smithers site were located in 4 younger stands that were classified as having mature trees present and 35% of nest trees were in stands adjacent to old-growth stands or younger stands with mature trees (Cotton 1999). This further supports the conclusion that, although flexible in their nest-site selection, flying squirrels seek out areas with larger trees. Ecosystem maps closely approximated actual ecosystem types at the Smithers site.

Northern flying squirrels exhibited a remarkable flexibility in nest-tree selection. Use of many relatively small trees for nest trees suggests that nests may exist in more situations than previously reported and suitable nest trees are not readily obvious based solely on size and condition of the tree. Although flying squirrels in our study did not appear to select particular habitat features relative to nest-site location, retaining large trees and structural diversity is likely important for the persistence of this species. That diversity may be more important than any particular attribute of the stand and should be the focus of further investigation because structural features such as large-diameter trees, snags, live cavity trees, and witches' brooms tend to be reduced in managed forests. Large, old trees, both live and dead, provide potential nesting structures, and coarse woody debris on the ground provides a substrate for forage production, including mushrooms. Structural diversity also may be important for movements of animals within the stand and for cover from predators (Harmon et al. 1986).

Current practices of patch retention in commercial harvesting (typically retaining 5–20% of the forested area of a cut block) may provide habitat for forest specialists by maintaining structural diversity within stands (Coates and Steventon 1995). Postharvest patches, however, may not be large enough to be used by flying squirrels until the surrounding 2nd-growth stand has reached a suitable age for travel and foraging. Flying squirrels are highly arboreal and are not likely to cross large openings that would require travel on the ground (Mowrey and Zasada 1984). Flying squirrels that recolonize remnant patches after surrounding stands have regenerated could potentially assist in the rebuilding of the mycorrhizal community in the cut area by dispersing fungal spores and transporting lichen fragments to the younger stand (Fogel and Trappe 1978). Sufficient mature forest, however, must remain in the landscape to sustain populations and provide for dispersal of flying squirrels.

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Climate Change and Forest Disturbances

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Climate Change and Forest Disturbances

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Studies of the effects of climate change on forests have focused on the ability of species to tolerate temperature and moisture changes and to disperse, but they have ignored the effects of disturbances caused by climate change (e.g., Ojima et al. 1991). Yet modeling studies indicate the importance of climate effects on disturbance regimes (He et al. 1999). Local, regional, and global changes in temperature and precipitation can influence the occurrence, timing, frequency, duration, extent, and intensity of disturbances (Baker 1995, Turner et al. 1998). Because trees can survive from decades to centuries and take years to become established, climate-change impacts are expressed in forests, in part, through alterations in disturbance regimes (Franklin et al. 1992, Dale et al. 2000).

> CLIMATE CHANGE CAN AFFECT FORESTS BY ALTERING THE FREQUENCY, INTENSITY, DURATION, AND TIMING OF FIRE, DROUGHT, INTRODUCED SPECIES, INSECT AND PATHOGEN OUTBREAKS, HURRICANES, WINDSTORMS, ICE STORMS, OR LANDSLIDES

Disturbances, both human-induced and natural, shape forest systems by influencing their composition, structure, and functional processes. Indeed, the forests of the United States are molded by their land-use and disturbance history. Within the United States, natural disturbances having the greatest effects on forests include fire, drought, introduced species, insect and pathogen outbreaks, hurricanes, windstorms, ice storms, and landslides (Figure 1). Each disturbance affects forests differently. Some cause large-scale tree mortality, whereas others affect community structure and organization without causing massive mortality (e.g., ground fires). Forest disturbances influence how much carbon is stored in trees or dead wood. All these natural disturbances interact with human-induced effects on the environment, such as air pollution and land-use change resulting from resource extraction, agriculture, urban and suburban expansion, and recreation. Some disturbances can be functions of both natural and human conditions (e.g., forest fire ignition and spread) (Figure 2).

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Figure 1. The major disturbance impacts on forests result from fire, drought, introduced species, insect and pathogen outbreaks, hurricanes, windstorms, ice storms, and landslides. Photo: Virginia Dale

Each disturbance has both social and economic effects (Table 1). Estimating the costs of each of these disturbances is very difficult; these estimates for the United States are illustrative only. Of the eight forest disturbances considered, ice storms are the least costly, averaging about $10 million and more than 180,000 ha annually (Michaels and Cherpack 1998). Insects and pathogens are the most expensive, with costs exceeding $2 billion and 20.4 million ha per year (USDA 1997). The socioeconomic aspects of these damages are only part of the cost. Costs of impacts to ecological services (e.g., water purification) can be large and long term.

This article examines how eight disturbances influence forest structure, composition, and function and how climate change may influence the severity, frequency, and magnitude of disturbances to forests. We focus on examples from the United States, although these influences occur worldwide. We also consider options for coping with disturbance under changing climate. This analysis points to specific research needs that should improve the understanding of how climate change affects forest disturbances.

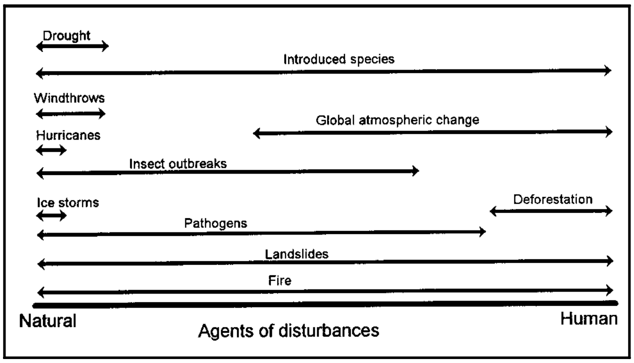

Figure 2. Natural and anthropogenic agents of forest disturbances that result from climate change (modified from Dale et al. 1998a). The length and position of the arrow relates to the extent of natural versus anthropogenic influence on the agent.

This paper is one in a series developed by the forest sector of the US National Assessment of the Potential Consequences of Climate Variability and Change. In examining how forests may be affected by climate change, the Forest Sector Committee divided the topic into four areas (processes, diversity, disturbances, and socioeconomics), each of which is the focus of an article in this issue of *BioScience*. Impacts of climate changes on aquatic disturbances are critical, but this paper focuses on direct terrestrial impacts. The effects of a rise in sea level, coastal processes, and salinity on terrestrial systems are examined in the coastal sector of the national assessment (NAST 2000).

Past and future climates in the United States

The Earth has experienced cycles of temperature and precipitation change on a geological scale, but recent evidence points to a large anthropogenic component to current global climate changes (Houghton et al. 1996). Analyses of the last 100 years of climate data for the coterminous United States suggest that the average temperature has risen by 0.5°C and that precipitation has increased 5%–10% (NAST 2000); observations also indicate that there has been some increase in precipitation and temperature extremes (Easterling et al. 2000). To look at future climates, scenarios from two of the newer, transient general circulation models (GCMs)—one developed by the Hadley Center in the United Kingdom (HADCM2SUL) and one by the Canadian Climate Center (CGCM1)—have been selected for this national assessment (MacCracken et al. 2000). These transient GCMs simulate atmospheric dynamics under a gradual increase in greenhouse gas concentrations from about 1895 to 2100 and produce scenarios (precipitation patterns, temperature changes, and so on) that forest-process and biogeography models use to examine transient community and ecosystem dynamics under climate change (Aber 2001, Hansen et al. 2001).

These two climate scenarios present a useful contrast for future climates. The HADCM2SUL produces relatively modest temperature increases over the United States (approximately 2.6°C) and large precipitation increases (about 20%); the CGCM1 simulates larger temperature increases

Table 1. Relative areal extent and economic cost of current disturbances in the United States.

Disturbance	Average annual impact area (ha)	Average annual economic cost (millions of dollars)
Fire	450,000[a]	261[b]
Hurricane	1,200,000[c]	700[d]
Tornado	450,000[e]	154[f]
Ice	>180,000[g]	>10[g]
Insects and pathogens	20,400,000[h]	1,500[i]
Exotic species	Nationwide	60[j]
Landslide	100,000	1,000[k]
Drought	Nationwide	Severity dependent

[a] Data from Ruiz (1996).

[b] From 1989 to 1994, fires destroyed 454,000 ha of US forests each year (Ruiz 1996). In 1994, the United States had 661,000 ha of forest fires with a total loss of $380 million, or $575 per ha burned. We assume that the geographic distribution of the 1994 fires represents the average distribution of fires.

[c] Based on the 1.8 million ha of South Carolina forest destroyed by Hurricane Hugo in 1989 and on the fact that an average of 0.67 major hurricanes per year struck the US mainland from 1900 to 1996 (Hebert et al. 1996).

[d] Obtained by multiplying the $700 in annual damage that occurs per year (Marsinko et al. 1997) by a 0.67 annual frequency.

[e] From Fujita (1971), we calculate an average area of damage to be 975 ha, multiply this value by the number of forest tornadoes in each region, and sum over all regions to obtain a first-order approximation of the total annual damage to forests by tornadoes.

[f] Across the southern United States, average harvest rotation length is 30 years, while across the North and Rocky Mountain region it is 70 years. Tornadoes destroy both the current year and accumulated previous years' growth. Annual returns of forestland range from $2.68 per ha in the Rocky Mountains to $23.46 per ha in the South (USDA 1990). Given that tornadoes affect all forest age classes, tornadoes destroy 35 years' worth of growth in the North and Rocky Mountains while destroying 15 years' worth of growth in the South. Assuming that the age classes are equally distributed and that downed timber is not salvageable, the total annual impact of tornadoes is approximately $154 million.

[g] Based on January 1998 ice storm damage across New England, with a 100-year frequency (Michaels and Cherpack 1998).

[h] The regional extent of insect- and pathogen-related forest damage is 20.4 million ha (USDA 1997). However, not all of the trees within this forested area are destroyed. Instead, insects and pathogens within this region annually kill some trees while reducing productivity for many others. Major insect pests include the southern pine beetle (3.0 million ha), gypsy moth (up to 2.6 million ha), other spruce and pine beetles (up to 1 million ha), and hemlock woody adelgid (areal extent unknown). Major pathogens include dwarf mistletoe (11.7 million ha), fusiform rust (about 1.8 million ha), white pine blister rust (areal extent unknown), and anthracnose (areal extent unknown).

[i] CP Harausz, personal communication, 2000.

[j] From Kräuchi (1993).

[k] From Schuster (1996).

(approximately 5.0°C) and similar model precipitation increases over the coterminous United States in the next 100 years (NAST 2000). The ecological models associated with the national assessment incorporate fire and drought disturbances, and we report the implications of these climate scenarios on these disturbances. The technology to incorporate other disturbances, such as windstorms or invasive species, is only now emerging. Therefore, the analyses we present here are based on new technology or are simply our best inference based on ecological models, literature surveys, or our professional judgment.

Climate influences on forest disturbances

A review of how each disturbance is influenced by climate, affects forests, and might be exacerbated by climate change provides a background for examining ways to cope with the impacts of climate change. The effects of each disturbance are partly tempered by prior adaptations. For example, species present in a forest reflect past disturbances. Droughty sites typically support species that survive well under dry conditions with uncertain rainfall. Sites that have frequent fires contain gymnosperm species with serotinous cones. Thus, if climate change alters the distribution, extent, frequency, or intensity of any of these disturbances, large impacts (such as loss of species regeneration) could be expected. The effects on species or communities already at the margin of their range may be particularly severe.

Fire. The frequency, size, intensity, seasonality, and type of fires depend on weather and climate in addition to forest structure and composition. Fire initiation and spread depend on the amount and frequency of precipitation, the presence of ignition agents, and conditions (e.g., lightning, fuel availability and distribution, topography, temperature, relative humidity, and wind velocity).

Fire effects on forests include acceleration of nutrient cycling, mortality of individual trees, shifts in successional direction, induced seed germination, loss of soil seed bank, increased landscape heterogeneity, changes in surface-soil organic layers and underground plant root and reproductive tissues, and volitalization of soil nutrients (Whelan 1995). Erosion can occur where soil disturbance accompanies fire (e.g., during fire fighting or timber salvage operations). Fire affects forest value for wildlife habitat, timber, recreation, and, through smoke, human health.

The rapid response of fire regimes to changes in climate (Flannigan et al. 1998, 2000, Stocks et al. 1998) can potentially

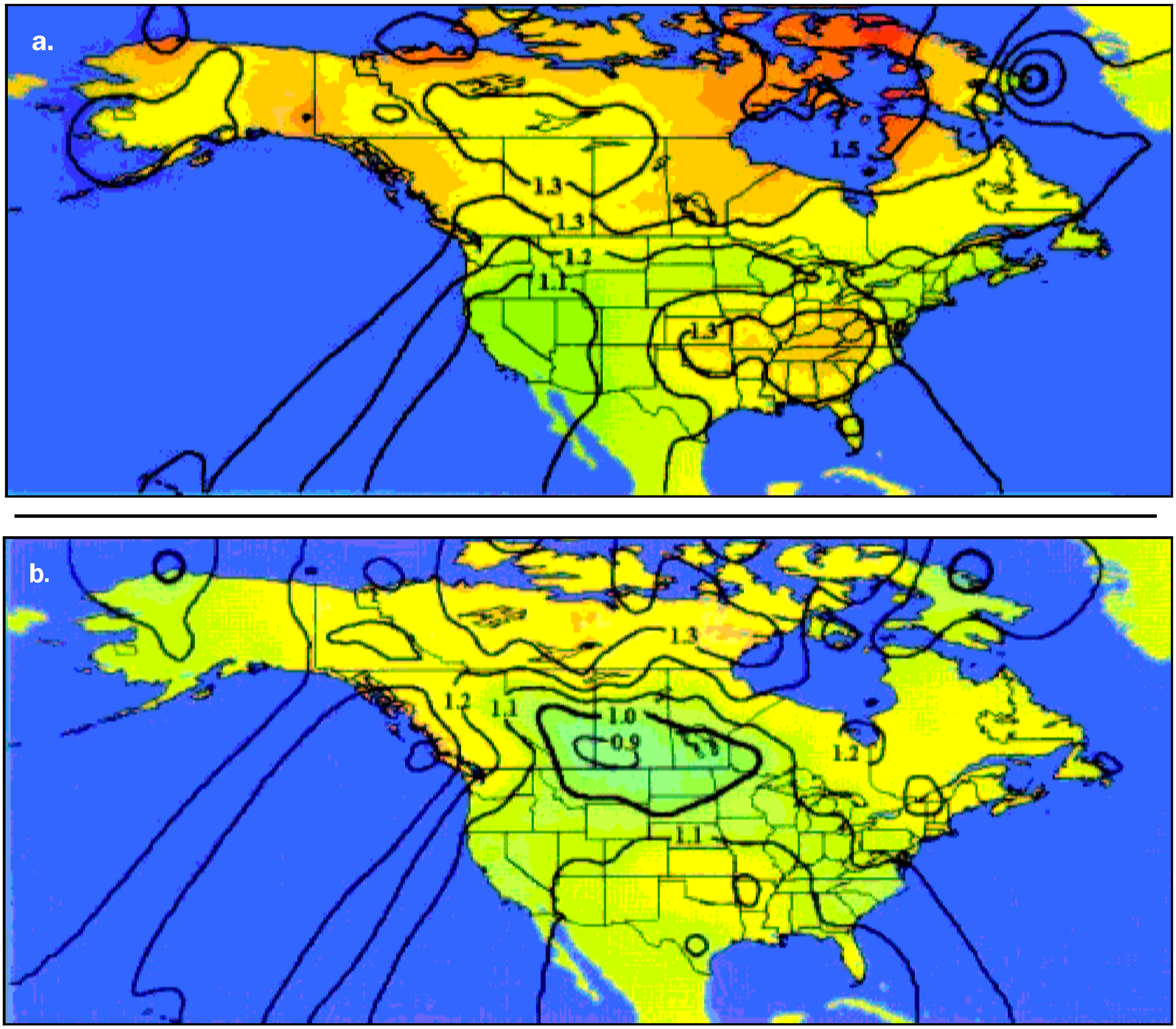

Figure 3. The ratio of the mean seasonal severity rating (SSR) between 2060 and the present day using (a) the Canadian GCM and (b) the Hadley GCM in the fire model described by Flannigan et al. (1998). The figures are a ratio of the future divided by the present, so that isolines of 1.0 mean no change, ratios greater than 1.0 mean an increase in SSR, and ratios less than 1.0 mean a decrease in SSR. The SSR is a measure of the fire weather severity and is a rough indicator of area burned. The average SSR for 1985 to 1994 was used for the present value, and an average for 2055–2064 for the 2060 value.

overshadow the direct effects of climate change on species distribution and migration. Modeling results predict great variation in future fire–weather patterns for the northern portion of North America (Figure 3). The seasonal severity rating (SSR) of fire hazard increases over much of North America under both the HADCM2SUL and the CGCM1 scenarios. The wetter Hadley scenario produces some small decreases in SSR for the Northern Great Plains, and increases are generally less than 10% over most of the rest of the continent. Some fire history studies suggest that the frequency of fire can decrease despite warmer temperatures because of increased precipitation (e.g., Bergeron and Archambault 1993). The warmer and drier CGCM1 produces a 30% increase in SSR for the southeastern United States and Alaska, with about 10% increases elsewhere. These scenarios suggest an increase in fire intensity and a 25%–50% increase in the area burned in the United States. In addition, recent results from the MC1 model, which is described by Neilson and Drapek (1998), show an increase in area and biomass burned under both scenarios. This model includes an interaction with CO_2 concentrations, which, through increased CO_2 fertilization and increased water-use efficiency, produces more biomass and thus more fuel, contributing to more and larger fires under a highly variable climate that has dry years interspersed with wet periods.

Drought. Droughts occur in nearly all forest ecosystems. Drought effects are influenced by soil texture and depth; exposure; species present; life stage; and the frequency, duration, and severity of drought. Droughts occur irregularly in forests of the humid regions east of the Mississippi River and in the superhumid Pacific Northwest. Droughts occur annually at the end of the growing season in forests at the midcontinental prairie–forest border, where annual precipitation ranges from 600–1000 mm, or within humid regions that have shallow or rocky soils. Seasonal summer droughts are experienced by western interior dry forests that depend on winter precipitation, such as forests in the semiarid plains and intermountain regions of the western United States. In some regions, droughts last several years.

The primary immediate response of forests to drought is to reduce net primary production (NPP) and water use, which are both driven by reduced soil moisture and stomatal conductance. Under severe conditions, plants die. Small plants, such as seedlings and saplings, are usually the first to die and can succumb under moderate conditions. Deep rooting and stored carbohydrates and nutrients make large trees susceptible only to severe droughts. Secondary effects also occur. When reductions in NPP are extreme or sustained over multiple growing seasons, increased susceptibility to insects or disease is possible, especially in dense stands (Negron 1998). Drought can also reduce decomposition processes, leading to a buildup of organic matter on the forest floor that may increase fire frequency or intensity or reduce nutrient cycling.

The consequences of drought depend on annual and seasonal climate changes and on whether the current drought adaptations are sufficient to confer resilience to new conditions (Hanson and Weltzin 2000). Forests tend to grow to a level of maximum leaf area that nearly fully uses soil water during the growing season (Neilson and Drapek 1998). A small increase in growing-season temperature could increase evaporative demand, triggering moisture stress. New results from two models described by Daly et al. (forthcoming), MAPSS and MC1, suggest that this mechanism may cause future increases in drought stress in the Southeast, southern Rockies, and parts of the Northwest. The MC1 model indicates that the Prairie Peninsula and Great Lakes region, parts of the Northwest, and the Gulf Coast could experience drought stress within two decades, even though these regions may become wetter in later decades.

Insect and pathogen outbreaks. Climate influences the survival and spread of insects and pathogens directly, as well as the susceptibility of their forest ecosystems. Changes in temperature and precipitation affect herbivore and pathogen survival, reproduction, dispersal, and distribution. Indirect consequences of disturbance from herbivores and pathogens include elimination of nesting trees for birds and negative effects on mycorrhizal fungi (Gehring et al. 1997, Ayres and Lombardero 2000). Other indirect effects include the impacts of climate on competitors and natural enemies that regulate the abundance of potential pests and pathogens.

Changes in the intensity and frequency of herbivore and pathogen damage in forests can have a range of effects. Most tree species support a community of other organisms, so the loss of any tree species can significantly reduce overall biodiversity. Such a loss occurred when chestnut blight almost completely eliminated chestnut trees (Opler 1979); the die-off of Fraser fir (*Abies fraseri*) caused by balsam woolly adelgid (*Adelges piceae*) also raises concerns. Herbivore and pathogen damage to trees can increase understory plant diversity (Stone and Wolfe 1996); the overall abundance and diversity of birds (Bennetts et al. 1996); and the diversity of predators, parasitoids, and detritivores (Savely 1939).

Because climate change can both directly and indirectly affect herbivores and pathogens through various processes, the ultimate effects on patterns of disturbance include increased disturbance in some areas and decreased disturbance in others. For example, an increase in the interannual variation in minimum winter temperatures is expected to favor more northerly outbreaks of southern pine beetles but could reduce more southerly outbreaks (Ungerer et al. 1999). Similarly, decreased precipitation and increased evapotranspiration should boost tree secondary chemical metabolism (and, therefore, resistance to pests) in forests that currently suffer modest growing-season water deficits (Reeve et al. 1995).

If global warming shifts species abundances, there may be associated shifts in herbivory. Compared to the cooler Paleocene, the Eocene had a greater diversity of herbivores and higher attack rates on the most abundant tree species (Wilf and Labandeira 1999). Increased warming would most likely increase the diversity of insects at higher latitudes. Because insects typically migrate much faster than trees, many temperate tree species are likely to encounter nonnative insect herbivores that previously were restricted to subtropical forests.

Introduced species. Introduced species can affect forests through herbivory, predation, habitat change, competition, alteration of gene pools via hybridization with natives, and disease (as either pathogens or vectors). Introduced species can alter the diversity, nutrient cycles, forest succession, and fire frequency and intensity of some ecosystems. The effects of introduced species should be considered concurrently with changes in native species distribution and abundance that occur as a consequence of climate change (Hansen et al. 2001). The impact of introduced species on ecosystems is influenced by such climatic factors as temperature, drought, and cloud cover (Ayres 1993). Invasion biology is not yet adept at forecasting impacts of invasions (Williamson 1999). The complex interactions among introduced species, native communities, managed and intensely harvested forests, and climate change compound this forecasting problem (Simberloff 2000).

The ultimate ranges of introduced species are largely determined by climate and human activities. Climate change will modify the distributions of many introduced species. Developmental rates will be modified by temperature change. For example, laboratory studies of balsam woolly adelgid growing under various temperature conditions provided the basis for simulations that suggest that temperature-induced changes in the population dynamics of the insect significantly affect Fraser fir survival (Dale et al. 1991).

The great majority of introduced species do not survive (Williamson 1999). Many fail because the climate is unsuitable at their points of arrival. Thus, a changed climate will lead to a different mix of surviving and failing species. In general, one might expect a larger fraction of survivors when the climate is warmer; introduced species comprise a far larger fraction of the biota in the warmer areas of the United States (Simberloff 1997).

Increased CO_2 can directly influence introduced plants through enhanced photosynthesis, but at different rates for different species. Resistance of trees to introduced herbivores is sensitive to both climate and CO_2 concentrations. Climate change, in concert with CO_2 concentration and nitrogen deposition, affects leaf nitrogen, which in turn influences herbivory.

Hurricanes. Hurricanes disturb forests of the eastern and southern coastlines of the United States, as well as those of the Caribbean islands and the Atlantic coast of Central America. Ocean temperatures and regional climate events influence the tracks, size, frequency, and intensity of hurricanes (Emanuel 1987). An average of two hurricanes make land every 3 years in the United States (Hebert el al. 1996). Global warming may accelerate the hydrologic cycle by evaporating more water, transporting that water vapor to higher latitudes, and producing more intense and possibly more frequent storms (Emanuel 1987, Walsh and Pittock 1998). However, other variations may override possible increases in hurricane frequency (Lighthill et al. 1994).

Changes in the global hydrologic cycle and temperature will influence hurricane formation, but we cannot yet predict the direction and magnitude of change. Sea-surface temperatures are expected to rise, with hotter temperatures expanding to higher latitudes (Royer et al. 1998, Walsh and Pittock 1998). Most studies point to an increase in hurricane frequency (Royer et al. 1998). However, even if frequency does not increase, it is likely that intensity and possibly duration of individual storms will increase because of the warming of the air and ocean, sources of energy for a hurricane (Emanuel 1987, Walsh and Pittock 1998).

The effects of hurricanes on vegetation include sudden and massive tree mortality, complex patterns of tree mortality (including delayed mortality), and altered patterns of forest regeneration (Lugo and Scatena 1996, Lugo 2000). These changes can lead to shifts in successional direction, higher rates of species turnover, and opportunities for species change in forests, which can in turn increase landscape heterogeneity, produce faster biomass and nutrient turnover, and result in lower aboveground biomass in mature vegetation (Lugo and Scatena 1995). Hurricanes can also result in buried vegetation and carbon sinks.

Windstorms. Small-scale wind events are products of mesoscale climatic circumstances and thus may be affected by climate changes, although the type and amount of alteration in windstorm characteristics cannot be predicted because these smaller-scale events are below the resolution of today's GCMs. Yet, tornadoes, downbursts, and derechos (a series of storm cells along a squall line) are probably the most important agents of abiotic disturbance to eastern deciduous forests (Peterson 2000). These disturbances can create very large patches of damage: A windstorm on 4 July 1999 in the Boundary Waters Canoe Area of Minnesota flattened roughly 250,000 acres of forest (Minnesota Department of Natural Resources press release, 12 July 1999). Windstorms can cause heavy mortality, produce canopy disruption, reduce tree density and size structure, and change local environmental conditions. Consequently, the disturbance may trigger advance regeneration, seed germination, and accelerated seedling growth (Peterson and Pickett 1995). These effects can change successional patterns, gap dynamics, and other ecosystem-level processes. The relationship between wind strength and severity of disturbance is not constant across different forests and species; although shallow-rooted species and thinned stands may be especially vulnerable to wind events, multiple factors influence tree response to high winds.

Berz (1993) suggests that increased intensity of all atmospheric convective processes will accelerate the frequency and intensity of tornadoes and hailstorms. Consistent with this view, Karl and colleagues (1995a) found that the proportion of precipitation occurring in extreme thunderstorm events increased in the United States from 1910 to 1990, and Karl and colleagues (1995b) further suggest that the climate of the United States has become more extreme (in terms of temperature and precipitation anomalies) in recent decades. Thus, it appears that the thunderstorm conditions that contribute to tornado formation have increased and are likely to continue increasing under projected climate changes. Furthermore, Etkin (1995) found a positive correlation between monthly tornado frequency and mean monthly temperature in western Canada, and inferred that this relationship suggests increased tornado frequency under a warmer climate scenario. Despite these inferences about tornado frequencies and the direct data on thunderstorm trends, understanding of tornado genesis is still inadequate to allow a direct forecast of how climate change will affect the frequency or severity of windstorms in the next century (Chuck Doswell [National Severe Storms Laboratory], personal communication, 2000).

Ice storms. Ice storms are caused by rain falling through subfreezing air masses close to the ground; those air masses supercool the raindrops, which freeze on impact. Ice accumulation can vary dramatically with topography, elevation, exposure, and areal extent of the region over which conditions favor glaze formation. Ice storms occur throughout the United States except along the southwestern borders and parts of the plains, but the frequency and severity of ice storm events increase toward the northeastern US borders. However, the historic record of ice-storm events over large areas has not been consistent or precise, with rigorous measurements of ice accumulation.

Ice storms affect trees, forests, and forested landscapes in different ways. Ice damage to trees can range from severing a few twigs, to bending stems, to moderate crown loss, to outright breakage of trunks. Depending on stand composition, amount and extent of ice accumulation, and stand history, damage to stands can range from light and patchy to total breakage of all mature stems (Irland 1998). Effects on forest stands include shifts in overstory composition in favor of more resistant tree species, loss of stand growth until leaf area is restored, and damage to stem form (Irland 2000). Damaged stems are then more susceptible to the impacts of insects and disease (Smith 2000). Recently thinned stands can be highly vulnerable because crowns have spread into the new space but branch strength has not developed. Several tree species can survive within areas frequented by ice storms. Though weather conditions producing ice storms are well understood, it is unclear how changes in climate will affect their frequency, intensity, regional location, or areal extent. However, atmospheric warming will most likely shift the locations of prevailing ice storms northward.

Landslides. Both slow and rapid movements of soil, rock, and associated vegetation are triggered directly by climate factors and indirectly by climate-influenced processes (e.g., stream-bank erosion) and by nonclimate factors such as earthquakes and volcanism. Triggering climatic events include snowmelt and intense rainfall, including that associated with hurricanes. Landslide frequency and extent are influenced by precipitation amount and intensity; snow accumulation, melt rate, and distribution; and roads and other land uses. The potential for a site to slide is influenced by slope steepness, properties of soil and rock, and hydrologic factors. Vegetation influences the likelihood of sliding through the soil-stabilizing effects of root systems and the effects of vegetation structure and composition on hydrology. Landslides remove soil and vegetation from steep slopes and damage forests on gentler slopes where landslide deposits come to rest. Landslides in forest landscapes can also damage aquatic resources and threaten public safety. Yet it is important to recognize that landslides are natural components of terrestrial and aquatic ecosystems.

Climate-change effects on landslides reflect changes in the delivery of water to soils through altered precipitation and snow hydrology (Buma and Dehn 1998). Because climate change is expected to vary geographically and with elevation, landslide responses will vary with similar complexity. Landslides are expected to be less frequent in areas where GCM scenarios predict reduced overall precipitation or reduced snowmelt because of warming trends, limiting snow accumulation (Buma and Dehn 1998, Dehn forthcoming). In the Pacific Northwest, much of the small, rapid landsliding occurs during rain-on-snow events in a broad elevation band where snow accumulates and melts several times in an average year. A simple warming without change in overall annual precipitation would be expected to result in reduced sliding by limiting the amount of snow (and its associated snowmelt) available to augment the rainfall reaching the soil. The most socially and ecologically significant landslides are triggered by intense precipitation. Thus climate change that increases storminess, and hence soil saturation, will increase landslide occurrence.

Interactions among disturbances. Many disturbances are cascading. Drought often weakens tree vigor, leading to insect infestations, disease, or fire. Insect infestations and disease promote future fires by increasing fuel loads, and fires promote future infestations by compromising tree defenses. Increased fire intensity or extent would enhance the potential for landslides. Also, changes in land use, forest management, and atmospheric chemistry can interact with these natural disturbances. For example, harvest and road establishment in landslide-prone areas coupled with increased wetness could result in more landslides. In the southern Appalachians, ozone exposure coupled with infestations of exotic insects and climate change may increase Fraser fir mortality and red spruce stress. In some cases, however, the combination of disturbances may ameliorate impacts. Under droughty conditions, stomata tend to close, reducing the effects of high ozone exposure.

Nevertheless, when ecosystems experience more than one disturbance, the compounded effects can lead to new domains or surprises (Paine et al. 1998). A new domain is entered when the system has not recovered from the first disturbance before a second perturbation occurs, leading the system to a new long-term condition. For instance, the combination of climatically driven wildfires, fragmentation caused by agricultural settlement, and logging in the boreal forest has resulted in significant and unprecedented changes in forest composition (Weir 1996). Invasive nonnative species are sometimes able to modify existing disturbance regimes or introduce entirely new disturbances (Mack and D'Antonio 1998). Under climate change, these compounded interactions may be unprecedented and unpredictable. They are likely to appear slowly and be difficult to detect because trees live for so long.

Strategies for dealing with forest disturbances

Coping strategies for forests are influenced by the value of the forest, the naturalness of the disturbance, and the range of acceptable management options. Often the least ecologically disruptive response after a disturbance is no action at all, but managers or society usually call for some type of cleanup or restoration, even when such action may retard recovery (Dale et al. 1998a). The value and management goals for the forest dictate how many resources can be allocated to its management. These values can change, as is illustrated by the revision of burn policy to recognize fire as a natural part of forest development that should not always be controlled.

Table 2. Coping strategies for dealing with disturbance effects on forests.

Managing the system before the disturbance

To reduce vulnerability:
- Altering forest structure (e.g., tree spacing and density, standing dead trees, or coarse woody debris on forest floor)
- Modifying the landscape structure (e.g., the size or location of management activity)
- Changing species composition (e.g., planting alternative species)

To enhance recovery:
- Altering structure (e.g., enhancing advance regeneration)
- Adjusting species composition (e.g., planting alternative tree species)

Managing the disturbance
- To reduce the opportunity for the disturbance to occur (e.g., regulating nonnative species introductions or use of fire)
- To reduce the impact of the disturbance (e.g., rapid response to control insects, pathogens, or fire)

Managing recovery
- To speed recovery (e.g., adding structural diversity, planting late-successional species, or reducing environmental stress)
- To reduce vulnerability to future disturbances (e.g., managing tree density, species composition, forest structure, and location and timing of management activities)

Monitoring for adaptive management
- To measure the state of the forest with and without disturbance
- To determine interactions between disturbances

The ability to manage for these eight disturbances varies greatly. However, current understanding of the disturbance nearly always provides some guidance for management under a future changed climate. Coping strategies for one disturbance type are often appropriate management responses to other disturbance types. For example, the removal of dead or dying trees and downed woody debris can reduce the risk of fire as well as alter insect and disease dynamics. Density management can reduce drought stress as well as alter insect population dynamics, but it could make forests more susceptible to wind. Thus, management effects are not always positive. Strategies for coping with disturbances in forests may also vary regionally. No matter where they are carried out, however, these practices often take 50–100 years to convert a landscape, and they are difficult to implement on inaccessible sites or in reserves.

We organize the coping strategies into several categories: managing before the disturbance, managing the disturbance itself, managing the recovery, and monitoring for adaptive management (Table 2). These options are presented independent of climate-change effects but with the understanding that climate may alter the disturbance regime.

Managing before a disturbance. Before a disturbance occurs, forests can be managed to reduce vulnerability or to enhance recovery. In both cases, management actions can alter the structure or the composition of the forest. In situations where the goal is to reduce the chance of future disturbances, adjustments to forest structure can be useful. For example, species or individual trees susceptible to ice or wind storms can be removed, as is common in cities. In addition, tree spacing and density can be altered to reduce susceptibility to drought. However, dead woody debris has numerous benefits (Harmon et al. 1986), and its extensive removal can affect the biota and nutrient cycling. Managers can also change species composition to reduce the vulnerability of forests to disturbances. Tree species that are less vulnerable to fire, droughts, wind, insects, or pathogens can be planted or maintained. For example, the colonization of phloem-feeding insects, such as bark beetles, is partially controlled by the ability of the tree to produce oleoresin, which is under genetic control. So, planting selected tree species and genotypes with relatively high oleoresin could limit insect outbreaks.

Landscape structural changes can also reduce the chances that future disturbances will damage the forest. The pattern of clear-cutting influences the potential for windstorms to blow down trees, because destructive winds are more prevalent along the edge of a cut (Savill 1993). And the placement of roads can influence the likelihood of future landslides and the spread of wildfire.

Management can be designed to reduce the opportunity for disturbance to occur. Examples are regulations that limit the introductions of nonnative species, the imposition of burning restrictions, and the use of controlled burns to reduce fuel loads. Trees can be planted that are less susceptible to disturbance. Species that promote disturbances can be removed. Density of trees can be managed to reduce the potential for future insect outbreaks or storm damage. Finally, roads can be designed to reduce the potential for landslides.

Other management actions can enhance forest recovery. Forest structure can be modified to speed up the successional process in the event of a disturbance. Alternatively, species composition can be adjusted to promote recovery. For example, in areas likely to experience a disturbance, trees with salvage value can be planted.

Managing the disturbance. Some disturbances, such as fire, insects, disease, and drought, can be managed during the disturbance through preventive measures or manipulations that affect the intensity or frequency of the disturbance. Alternatively, the disturbance can be managed to reduce its impact. A common way to control outbreaks of the southern bark beetle is to be on the alert for sites experiencing some beetle damage, then to cut those trees quickly to reduce the size of the area affected. Fire control is another example of a management action to reduce the impact of a disturbance.

Managing the recovery. Recovery efforts can focus either on managing the state of the system immediately after the disturbance (e.g., salvage logging) or managing the ongoing process of recovery (e.g., planting and reseeding). Recovery efforts need careful consideration of the long-term impacts because such actions can damage soils and residual trees. Stands can recover naturally without any removal of the dead or damaged trees.

Recovery actions can be designed to speed recovery. In the aftermath of a disturbance, recovery can be enhanced by

Table 3. Research questions about how disturbances affect forests in the face of climate change. (The numbers refer to the interactions indicated in Figure 4.)

1. Improved understanding of climatological conditions that initiate disturbances
- What are the average and range of climate-change predictions?
- What information about climate and weather forecasts are needed to improve both short- and long-term predictions of disturbance effects on forests?
- How do interactions between forest structure and function and climate affect disturbances?
- How does climate variability interact with the temporal and spatial variability of forest disturbances?

2. Better information on how disturbances and land-use changes affect climate
- How do changes in forest structure caused by disturbance influence weather and climate?
- Can hurricanes transport enough heat and moisture to alter climate?

3. Quantifying the impacts of disturbances on forests
- What are the average and range of the frequency, intensity, and spatial extent of forest disturbances?
- What are the major environmental factors affecting forest disturbance regimes?
- What are the major impacts of disturbance on forests?
- What patterns of species composition and yield are altered by disturbances (especially at the margin of species ranges)?
- What are the long-term effects of a disturbance, and how can they be quantified?

4. Interactions among forest disturbances and management
- What information is needed to understand the response of a forest to multiple disturbances?
- How do forest disturbances interact?
- What options exist for managing forests in the face of climate change?
- How should forests be monitored to best inform management of impending changes?

adding structural elements that create shade or other safe sites necessary for reestablishing vegetation or that serve as perches for birds (and thus places where seeds would be dispersed). Alternatively, late successional species can be planted to speed up succession. Finally, additions of water or nutrients can reduce environmental stress and facilitate restoration. Recovery can also be managed to reduce vulnerability to future disturbances.

Monitoring for adaptive management. A monitoring program should be used to determine how disturbances affect forests and to continually update our understanding of how climate change is potentially influencing the disturbance regimes. Monitoring can be designed to measure the state of the forest with and without disturbance under different management activities or to identify potential risks of forest disturbances. Such information is used to inform management of the potential outcomes of management actions.

Although many coping strategies associated with these disturbances could be incorporated into current forest-management practices regardless of climate change, the potential changes in climate may create a novel disturbance. For example, climate change may allow the migration of nonindigenous species into a forest, and current understanding of interactions and coping strategies may not apply to the resulting competitive interactions between nonindigenous and native species. Adaptive management approaches management as a continual learning process (Walters 1986). The continued monitoring of ecosystem structure and function could be part of the coping strategy to address the likely surprises. The impacts of insects and pathogens are already monitored through the Forest Health Program, and weather and fuel moisture are monitored to assess the risk of fire during the fire season. However, few surveys quantify the extent and severity of damage from wind and ice storms or landslides.

Information from monitoring programs could be used to update risk assessments in management plans and prescriptions in an adaptive-management sense. A risk-ranking system could identify aspects of the forest most susceptible to disturbance under a changing climate. In conjunction with spatially explicit modeling of the site under various scenarios of disturbance impacts, a risk map could be created to identify sites most in jeopardy (Dale et al. 1998b).

Research needs

A key feature of this analysis is the realization of our lack of knowledge in many critical areas. The numbered aspects of Figure 4 depict major interactions about which more information is needed. We determined the key research needs for each disturbance and then organized the questions that must be resolved into the six topics discussed below. Examples of broad research questions are given in Table 3. Such research will lead to better management decisions.

Understanding climatological conditions that initiate disturbance. Accurate projections of climate effects of disturbances require improved climate and weather forecasts. The projections should include not only average climate conditions but also their range and variance. Short-term weather forecasts will be needed to predict drought occurrences for existing forests. For long-term climate change projections, improved resolution in climate models is needed so that regional patterns can be projected.

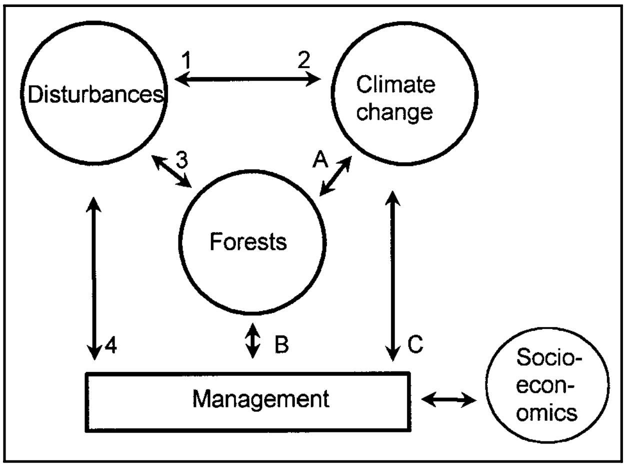

Figure 4. Interactions among disturbances, climate change, forests, and management strategies. The numbered arrows are the focus of research questions addressed in Table 3. The lettered interactions are covered in other analyses; A and B are discussed elsewhere in this issue of BioScience*, and C is discussed by Houghton et al. (1996). Management would include information from the social and political arenas as well as feedbacks from disturbances, climate change, and the forests themselves.*

We have limited understanding of what climatological conditions lead to some disturbances. Improved understanding of local meteorological events that spawn tornadoes is needed, as well as improved projections of conditions that foster thunderstorms. Our ability to predict the occurrence of fires and hurricanes has benefited from research that allows managers to focus their attention on sites most likely to be disturbed. However, some disturbances result from interactions between ecological and climatological conditions that are often poorly understood. For example, better monitoring is needed to improve the characterization of ice accumulation in relation to storm characteristics and associated weather, especially the delineation of areas by amount of ice accumulation. Once the relationship between climate and disturbances has been quantified, more-accurate predictions of disturbances can be developed to minimize their impact.

Understanding the effects of disturbances on microclimate. Because land-cover patterns can affect atmospheric circulation and cloud formation (Segal et al. 1988), changes in forest structure in the aftermath of fire, wind or ice storms, hurricanes, landslides, drought, and pest outbreaks may alter weather or climate conditions. This interaction needs to be studied and better understood.

Quantifying impacts of disturbances on forests. There is a paucity of basic information on the frequency, intensity, and spatial extent of some disturbances and their impacts on forests. This problem is especially severe for landslides, ice storms, and small wind events. For example, reconstructive studies should be done to determine the long-term influence of successive ice storms on forests. Such analysis also allows exploration of interactions between disturbances and delayed responses.

Research should identify herbivores and pathogens that are likely to be key agents of forest disturbance in the next 50 years. Integrated continental surveys are needed to determine the sensitivity of different types of pests and diseases to environmental change and the potential for increased outbreaks of insect herbivores and pathogens at the margins of their existing ranges.

Interactions between forest disturbances and management. Our ability to manage forests now as well as under climate change rests on our understanding of how forests respond to multiple disturbance events. A better understanding of interactions among fire, hurricanes, and biological disturbances (such as insects, pathogens, and introduced species) would improve our long-range predictions about forest succession and ecosystem dynamics and would lead to better prediction of conditions under which one event would affect the response to a subsequent one. This understanding, however, is complicated by the diverse goals of forest management (e.g., fiber products, wildlife habitat, biodiversity, and recreation).

Some management practices have been developed to cope with the physical disturbances of droughts, hurricanes, and wind events (Savill 1993). However, additional research could expand options for management. Research is needed on the mitigation of hurricane impacts (i.e., how to hurricane-proof landscapes and how to design protected areas, for example, determining what their area, shape, and distribution should be). Forest ecologists and land managers are exploring the prospects for tailoring forest management regimes to the range of ecosystem conditions and wildfire disturbance regimes observed in the past, and in some cases to those anticipated under future climate conditions. For drought, new field experiments could test forest sensitivity to specific climate-change projections in combination with changes in the concentration of atmospheric trace gases. How the genetic diversity of host plants will determine the future epidemiology of forest pathogens needs further exploration. Critical evaluations of known patterns of species change and yield following past climate changes are needed, along with models of succession that incorporate disturbance processes.

Conclusions

Over geologic time, changes in disturbance regimes are a natural part of all ecosystems. Even so, as a consequence of climate change, forests may soon face rapid alterations in the timing, intensity, frequency, and extent of disturbances. The number and complexity of climate variables related to forest disturbance make integrated research an awesome challenge. Even if changes cannot always be predicted, it is important to consider ways in which impacts to forest systems can be mitigated under likely changes in disturbance regimes. The task for the next decade is to understand better how climate affects disturbances and how forests respond to them. Improved monitoring programs and analytic tools are needed to develop this understanding. Ultimately, this knowledge should lead to better ways to predict and cope with disturbance-induced changes in forests.

Acknowledgments

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Imperiled Amphibians: A Historical Perspective

Chapter · January 1997

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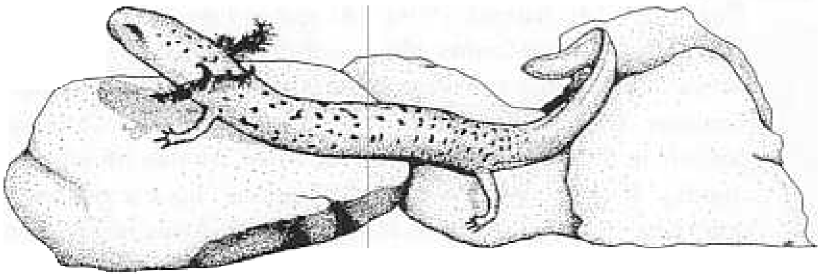

Imperiled Amphibians: A Historical Perspective

C. Kenneth Dodd, Jr.

Amphibians have been in and out of the news over the past few years because of the often unexplained disappearance of individual species or groups of species (Barinaga, 1990; Blaustein and Wake, 1990; Vitt et al., 1990; Wyman, 1990; Anonymous, 1991; Wake, 1991; Livermore, 1992, Blaustein et al., 1994c; Stebbins and Cohen, 1995). Amphibian declines or extinctions have been particularly apparent in the western United States (e.g., Bradford, 1991; Carey, 1993a; Fellers and Drost, 1993) and Australia (Richards et al., 1993), with scattered declines reported in Central and South America, Europe, and elsewhere (Vial and Saylor, 1993). Although much debate centers on natural population fluctuations (Pechmann et al., 1991; Blaustein, 1994; Pechmann and Wilbur, 1994) and their role in population viability (Sjogren, 1991a), there seems to be little doubt that amphibian populations are threatened by an ever expanding human population.

In this paper, I present an overview of the taxonomic diversity of the amphibians of the southeastern United States, the types of habitats used by amphibians, amphibian life history in relation to aquatic habitats, the types of studies that have been conducted on southeastern amphibians, and the status of and threats to particular species and populations. I define the southeastern United States to include an area from Virginia, West Virginia, and Kentucky, south through Florida and west to eastern Texas. As such, the region includes the contiguous southern Appalachians, southeastern Coastal Plain, Interior Highlands, and Edwards Plateau, all areas of important species richness and diversity.

TAXONOMIC REVIEW

Of the estimated 4,300 to 4,500 amphibian species worldwide (Vial and Saylor, 1993;

Aquatic Fauna in Peril: The Southeastern Perspective, edited by George W. Benz, and David E. Collins. 1997. Special Publication 1, Southeast Aquatic Research Institute, Lenz Design & Communications, Decatur, GA, 554 p.

Zug, 1993; McDiarmid, 1994), 147 species have been described United States. In addition to the described species, a number of sp nomic description, particularly in the salamander families Plethod Sirenidae (P. Moler, Florida Game and Fresh Water Fish Commiss possibly in the frog family Ranidae (R. Franz, Florida Museum o comm.). Within North America, the Southeast has the greatest a ness (Kiester, 1971). Most of the native amphibians in the Sou salamanders, with 99 described species. The amphibian species ric ern state is shown in Figure 1.

Order Caudata

Of the seven salamander families in the southeastern United Stat and Sirenidae) are endemic to the region while two additional fam and Proteidae) have their greatest species richness in the Southeas tant cryptobranchids occurs primarily in southern streams and ri species (*Andrias* spp.) are found in Asia. The family Plethodontic diversity in the Southeast, although its greatest species richness occ Neotropics of southern Mexico and Central America. The family S rily Palearctic and Oriental in distribution, although all th *Notophthalmus* are found in the Southeast.

The following salamander genera have their centers of distribut east: *Cryptobranchus*, *Necturus*, *Amphiuma*, *Siren*, *Pseudobranchus*, *Leu* *Haideotriton*, *Stereochilus*, and *Typhlomolge*. Most or all species of *Notop* *Eurycea*, *Gyrinophilus*, *Plethodon*, and *Pseudotriton* also occur in the ranges of individual species may extend substantially northward.

Order Anura

There are no endemic families of frogs in the southeastern Unite genera (*Acris* and *Pseudacris*) have centers of species richness with highest diversity (19 species) of southeastern frogs occurs within t followed by the Ranidae (true frogs: 14+ species) and the Bufonidae *Hypopachus* and *Syrrhophus* barely enter the coastal plain in Texas, and *Scaphiopus* are together represented by a total of five species, enter the Southeast. In addition to the 48 native frog species, at le (*Bufo marinus*, *Eleutherodactylus coqui*, *E. planirostris*, *Osteopilus se* tablished breeding populations (all in Florida).

DISTRIBUTION AND HABITATS

Physiographic Regions and Centers of Species Dive

Amphibians are found in all physiographic regions of the south (Table 1). They are found from sea level to the tops of the highes tains. Centers of species richness and endemism include the follo Mountains, particularly at higher elevations (salamanders, e

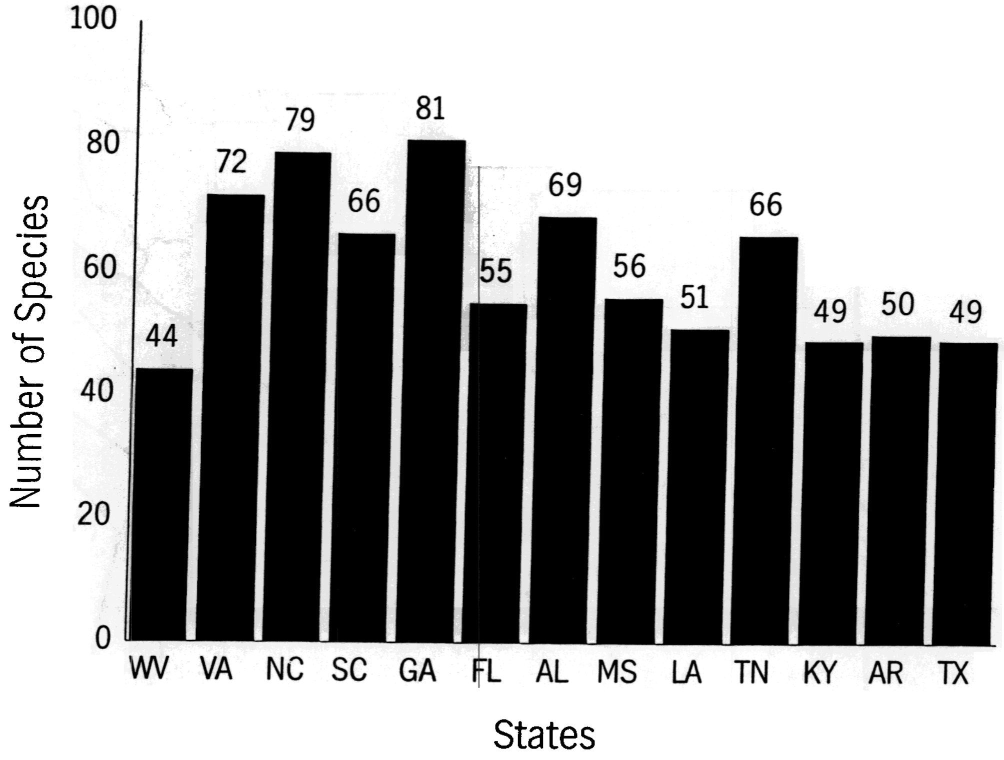

Figure 1. Species richness of amphibians in the southeastern United States. Data for Texas include only species found in the east Texas coastal plain.

Plethodontidae and the genus *Plethodon*); the Atlantic and Gulf of Mexico coastal plains (many salamanders and frogs, especially *Amphiuma*, *Siren*, *Pseudobranchus*, *Necturus*, *Haideotriton*, and *Pseudacris* species); the Interior Highlands, including the Boston, Ouachita, and Ozark mountains (many endemic salamanders); and the Edwards Plateau (many endemic cave and spring salamanders of the genera *Eurycea* and *Typhlomolge*) (Figure 2).

AQUATIC HABITATS

Amphibians are found in all types of aquatic wetlands (see Hackney et al., 1992, and references therein) except those associated with the saline waters along the coast. Even there, however, some species occasionally are found in brackish habitats (Neill, 1958; Christman, 1974). Selected references on amphibian species composition of southeastern aquatic environments include the following: Moler and Franz (1988), LaClaire and Franz (1991), Dodd (1992), and Cash (1994) for temporary ponds; Adams and Lacki (1993) for road-ruts; Turner and Fowler (1981) and Lacki et al. (1992) for ponds at former mine sites; Delis (1993) and O'Neill (1995) for wetlands in pine flatwoods; Mitchell et al. (1993) for saturated forested wetlands; Harris and Vickers (1984) and Vickers et al. (1985) for cypress domes; Pearson et al. (1987) for bayheads; Wright (1932), Delzell (1979), and Hall (1994) for large swamps; Dalrymple (1988) for wet prairies; Parker (1937) and Bancroft et al. (1983) for lakes; and Southerland (1986) for streams. Much information on amphibian use of aquatic habitats is contained in state or re-

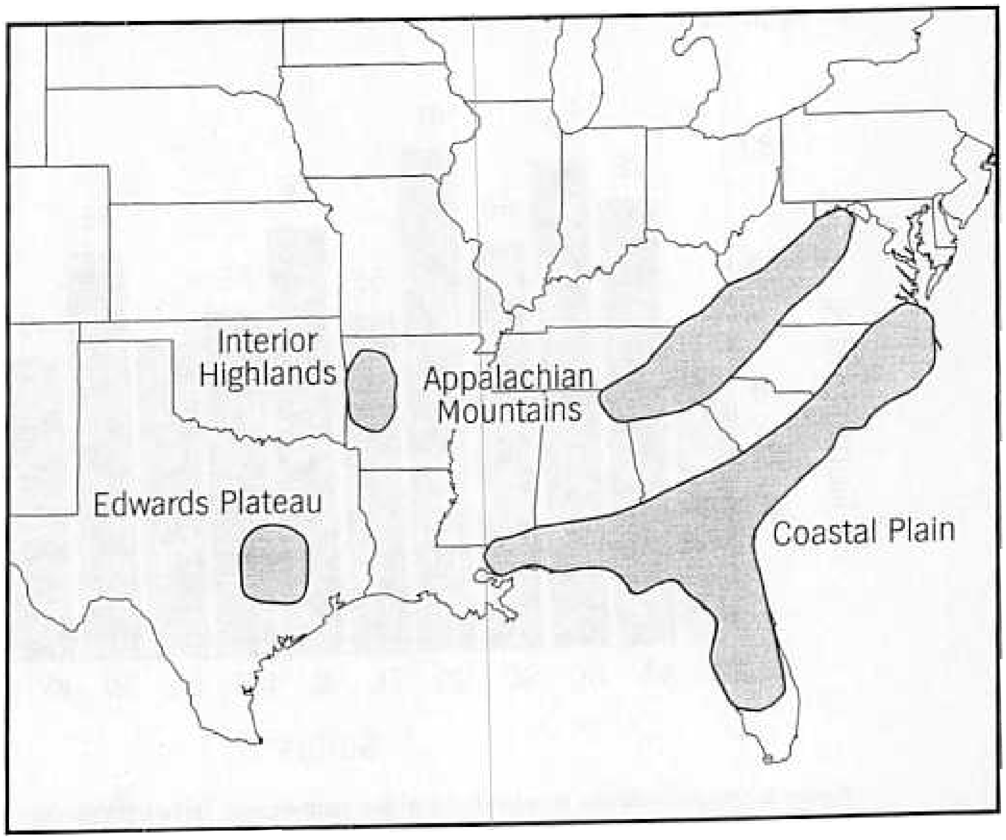

Figure 2. Centers of endemism and species richness (shaded areas) for the amphibian fauna of the southeastern United States.

gional books (e.g., Carr, 1940; Duellman and Schwartz, 1958; Ashton and Ashton, 1988; Dundee and Rossman, 1989; Gibbons and Semlitsch, 1991) as well as in numerous accounts of species in need of conservation (see Table 2).

Large, fully aquatic salamanders (*Cryptobranchus*, *Necturus*) typically are found in the larger rivers and streams, whereas small aquatic salamanders (*Desmognathus*, *Leurognathus*, *Eurycea*) frequent small streams and seeps. For these salamanders, larval development occurs within the stream and, after metamorphosis, adults live along the wet streamsides or among the gravelly substrate. Salamanders belonging to *Siren*, *Pseudobranchus*, and *Amphiuma* inhabit various types of vegetated ponds and mucky swamps. Newts and most *Ambystoma* species require temporary ponds to complete metamorphosis, and premature pond drying is an ever present threat to their development (Semlitsch, 1987; Pechmann et al., 1989; Dodd, 1993). Of course, even salamanders that do not require water to breed need moist environments to prevent desiccation.

As with salamanders, frogs use a variety of wetlands for reproduction. Most frog species have tadpoles which develop within ponds, lakes, wet prairies, or other lentic waters. Fewer species use streams, rivers, or swift flowing waters (e.g., *Rana heckscheri* in rivers, streams, and oxbows in addition to lentic waters). Some frogs are very habitat specific,

Table 1. Characteristics of the aquatic habitats and major ecosystems of native amphibians of the southeastern United States.[1]

Taxa	Number Species	Adult Habitat	Larval Habitat	Aquatic Habitats	Physiographic Provinces
Order Caudata — Salamanders					
Family Cryptobranchidae					
Cryptobranchus	1	A	A	R,LS	M,CUP,O
Family Proteidae					
Necturus	5?	A	A	R,LS,SS	
Family Amphiumidae					
Amphiuma	3	A	A	L,P,SW	
Family Sirenidae					
Pseudobranchus	2	A	A	L,P,SW	CP
Siren	2?	A	A	L,P,SW	CP
Family Ambystomatidae					
Ambystoma[2]	10	T,A	A	P	CP,P,M,O,CUP
Family Salamandridae					
Notophthalmus[2]	3	T,A	A	P	CP,P,M,CUP
Family Plethodontidae					
Desmognathinae					
Desmognathus	12	S,T	A,D	SS,SW	CP,P,M,O,CUP
Leurognathus	1	A	A	SS	M
Phaeognathus	1	T	D		CP
Plethodontinae					
Aneides	1	T	D		M,CUP
Eurycea	14?	A,T	A	SS,SW,C	CP,P,M,O,CUP,EP
Gyrinophilus	3?	S	A	SS,C	M,CUP
Haideotriton	1	A	A	C	CP
Hemidactylium	1	T	A	SW,SS	CP,P,M,O,CUP
Plethodon	33?	T	D		CP,P,M,O,CUP,EP
Pseudotriton	2?	S	A	SW,SS	CP,M,P,CUP
Stereochilus	1	A	A	SW	CP
Typhlomolge	2	A	A	C	EP
Typhlotriton	1	S	A	C,SS	O
Order Anura — Frogs					
Family Bufonidae					
Bufo	7	T	A	P,L,SW	CP,P,M,O,CUP,EP
Family Hylidae					
Acris	2	T	A	P,L	CP,P,M,O,CUP,EP
Hyla	8	T	A	P,L,SW	CP,P,M,O,CUP,EP
Pseudacris	9	T	A	P,L,SW	CP,P,M,O,CUP,EP
Family Microhylidae					
Gastrophryne	2	T	A	P	CP,P,O,CUP,EP
Hypopachus		T	A	P	CP
Family Leptodactylidae					
Syrrhophus	2	T	D		CP
Family Pelobatidae					
Scaphiopus	3	T	A	P	CP,P,O,CUP,EP

Table 1. Continued.

Taxa	Number Species	Adult Habitat	Larval Habitat	Aquatic Habitats	Physiographic Provinces
Family Ranidae					
Rana	14?	S,T	A	L,P,SW,R,LS,SS	CP,P,M,O,CUP,EP

[1] Adult habitat (T = terrestrial, S = semiaquatic, A = aquatic); Larval habitat (A = aquatic, D = direct development on land); Aquatic habitats (R = river, LS = large stream, SS = small stream, P = pond, L = lake, SW = swamp, bog or seep, C = cave); Physiographic provinces (CP = Coastal Plain, P = Piedmont, M = Appalachian Mountains, O = Ozark, Ouachita and Boston mountains, CUP = Cumberland Plateau, EP = Edwards Plateau). ? indicates that undescribed species are thought to be present.

[2] While most species have larvae that transform into adults, paedomorphic adults are not uncommon in some species or populations.

such as *Rana capito* and *Hyla gratiosa*, which require fishless temporary ponds for reproduction. Some species, such as *Bufo terrestris*, breed in a wide variety of wetland habitats.

TERRESTRIAL HABITATS

Although amphibians are usually associated with water, most species spend a substantial amount of time in terrestrial habitats. Individuals of some species often can be found at great distances from the nearest breeding ponds. For example, I have funnel-trapped many small frogs and salamanders in the harsh Florida sandhills 200 to greater than 800 m (656 to more than 2,624 feet) from the nearest water (Dodd, 1996). Franz et al. (1988) recorded a gopher frog (*Rana capito*) at a tortoise burrow 2 km (1.25 miles) from where the frog had been previously marked. Such long distance movements probably are not unusual. Greenberg (1993) captured southern toads (*Bufo terrestris*), eastern narrow-mouthed toads (*Gastrophryne carolinensis*), and eastern spadefoot toads (*Scaphiopus holbrooki*) in Florida sand pine scrub between 5 and 6 km (3.1 and 3.7 miles) from the nearest known water source.

Terrestrial refugia include caves (Saugey et al., 1988; Franz et al., 1994); burrows of tortoises (Jackson and Milstrey, 1989), pocket gophers, crayfish (especially by *Rana areolata* and *R. capito*) and other invertebrates; tree roots; rock crevices; surface debris; and probably many other subterranean habitats. Treefrogs often use arboreal retreats. Selected references on the use of terrestrial habitats by amphibians that require water to breed include Gibbons and Bennett (1974), Bennett et al. (1980), Semlitsch (1981), Campbell and Christman (1982), Pearson et al. (1987), Stout et al. (1988), Scott (1991), McCoy and Mushinsky (1992), and Dodd (1996).

AQUATIC AMPHIBIAN LIFE HISTORY

In North America, most amphibians have a biphasic life cycle consisting of an aquatic egg and larval stage, metamorphosis into a terrestrial adult, and migration back to water to breed and lay eggs. The time between metamorphosis and first breeding varies among species, although this period is usually one to four years (Duellman and Trueb, 1986). The life span of wild individuals also varies. For example, *Gastrophryne carolinensis* may live four or more years (Dodd, 1995a), whereas the hellbender may live greater than 25 years (Peterson et al., 1983). Generally, salamanders live longer than frogs, and larger species live longer than smaller species (Duellman and Trueb, 1986). Duellman and Trueb (1986) discussed life history variation and the factors that affect reproduction, life cycles, and other facets of amphibian biology.

There are exceptions to the "typical" amphibian life cycle. All non-hemidactyliine salamanders of the family Plethodontidae (i.e., *Aneides* and *Plethodon* species), two species of *Desmognathus* (*D. aeneus* and *D. wrighti*), and *Phaeognathus hubrichti* skip the aquatic larval stage (Table 1). Instead, eggs are laid on land in moist environments, the larval stage is passed within the egg, and the hatchling resembles a miniature adult.

Several salamanders, including all *Siren* spp., *Pseudobranchus* spp., *Necturus* spp., and *Typhlomolge* spp., some *Eurycea* spp., and *Haideotriton wallacei* and *Cryptobranchus alleganiensis*, are entirely aquatic and never leave the water or boggy wetlands. Eggs are deposited in vegetation, debris, or under rocks, young usually pass through a larval stage, and adults often retain larval features, such as exposed gills. *Amphiuma* spp. generally are aquatic, although eggs are deposited on land near water. Other species (*Ambystoma talpoideum*, *Notophthalmus* spp.) have individuals or populations that are facultative paedomorphs (that is, they become reproductively active while otherwise retaining larval phenotypes, and they never transform into adults while permanent water remains).

All native southeastern frogs, with the exception of the direct developing *Syrrhophus* spp., have a "typical" amphibian life cycle. Both of the introduced *Eleutherodactylus* spp. are direct developers with no aquatic life stage.

STUDIES AND LITERATURE ON IMPERILED SOUTHEASTERN AMPHIBIANS

Prior to the second half of the 20th century, most studies of the aquatic Amphibia of the southeastern United States focused on general distribution patterns, morphological systematics, and life history field observations. The literature stemming from those studies has been summarized in several major monographs and field guides (e.g., for Alabama see Mount, 1975; for Florida see Carr, 1940, and Ashton and Ashton, 1988; for Kentucky see Barbour, 1971; for Louisiana see Dundee and Rossman, 1989; for Texas see Dixon, 1987, and Garrett and Barker, 1987; for Virginia and the Carolinas see Martof et al., 1980; and for West Virginia see Green and Pauley, 1987). Books that will include extensive data on the biology of aquatic amphibians are in progress for the states of Tennessee and Virginia. In addition, separate herpetological bibliographies are available for the states of Florida (Enge and Dodd, 1992), Tennessee (Redmond et al., 1990), and Virginia (Mitchell, 1981).

Relatively recent concern for individual species has resulted in a series of books and journal articles which include a status assessment of actually or potentially imperiled aquatic amphibians. Reviews are available for West Virginia (Pauley and Canterbury, 1990), Virginia (Linzey, 1979; Pague and Mitchell, 1987; Terwilliger, 1991), North Carolina (Cooper et al., 1977), South Carolina (Harrison et al., 1979), Tennessee (Echternacht, 1980), Kentucky (Branson et al., 1981), Florida (McDiarmid, 1978; Moler, 1992d), Alabama (Mount, 1986c), and Arkansas (Reagan, 1974). Ashton (1976) provided a national checklist of amphibians and reptiles in need of conservation, and Bury et al. (1980) summarized

Table 2. Aquatic-dependent amphibians known or suspected of requiring conservation or management attention in the southeastern United States.[1]

Taxon	Threats — Rarity	States	Information Sources
Order Caudata — Salamanders			
Ambystoma cingulatum	habitat destruction	AL,FL,GA,SC	Ashton (1992); Palis (1992, 1993); M. Bailey, J. Palis, W. Seyle, J. McLemore (all pers. comm.); Means (1986c); Bury et al. (1980); Means et al. (1996)
A. mabeei	habitat destruction, forestry	VA	Pague and Mitchell (1991a)
A. maculatum	drought	GA	C. Camp (pers. comm.)
A. talpoideum	unknown, peripheral	AR,KY,NC,TN,VA	Echternacht (1980); Branson et al. (1981); Braswell (1977a); Pague and Mitchell (1991b); Reagan (1974); Trauth et al. (1993a)
A. texanum	rare?	AL	Folkerts (1986b)
A. tigrinum	locally rare, habitat destruction, exotics	AL,FL,VA	Mount (1986a); Travis (1992); Pague and Buhlmann (1991); M. Bailey (pers. comm.)
Amphiuma pholeter	rare?, siltation	AL,FL,GA	Bury et al. (1980); Means (1986d, 1992b)
A. tridactylum	unknown	KY	Branson et al. (1981)
Cryptobranchus alleganiensis	rare?, peripheral, habitat destruction, pollution, collecting	AL, AR, FL, GA, KY, MS NC, SC, TN, VA, WV	Redmond (1986); Pague (1991c); Echternacht (1980); Branson et al. (1981); Trauth et al. (1992, 1993b); Bruce (1977a); Bury et al. (1980); Reagan (1974); Nickerson and Mays (1973)
Desmognathus auriculatus	unknown	AL,FL,SC	Means (1986e); S. Christman, J. Harrison (both pers. comm.)

Table 2. Continued.

Taxon	Threats — Rarity	States	Information Sources
D. ochrophaeus	rare?	AL	Folkerts (1986c)
D. monticola	unknown, peripheral	AL,FL	Folkerts (1986a); Means (1992c)
D. quadramaculatus	peripheral, collecting	WV	Pauley and Canterbury (1990)
D. welteri	unknown	TN	Echternacht (1980)
Eurycea aquatica	habitat alteration	AL,TN	Bury et al. (1980)
E. longicauda	peripheral	NC	Bruce (1977b)
E. lucifuga	habitat destruction, collecting	WV	Pauley and Canterbury (1990)
E. multiplicata	habitat alteration, pesticides?	AR	Reagan (1974)
E. nana	habitat alteration	TX	Bury et al. (1980); USFWS (1984a)
E. soosorum	habitat destruction	~~TX~~	~~Chippindale et al. (1993)~~
~~*E. tridentifera*~~	rare?	TX	Bury et al. (1980)
E. tynerensis	habitat alteration	AR	Reagan (1974); Bury et al. (1980)
Gyrinophilus gulolineatus	habitat alteration, collection	TN	Echternacht (1980)
G. palleucus	unknown,habitat alteration	AL,TN	Simmons (1975); Ashton (1986); habitat alteration Echternacht (1980); Bury et al. (1980)
Haideotriton wallacei	rare	FL,GA	Bury et al. (1980); Means (1992d)
Hemidactylium scutatum	peripheral, unknown	AR,FL,KY,TN	Means (1992e); J. Palis (pers. comm.); Echternacht (1980); Branson et al. (1981); Reagan (1974); Saugey and Trauth (1991)
Leurognathus marmoratus	habitat destruction	VA	Gourley and Pague (1991)
Necturus sp.	habitat destruction	AL	Ashton and Peavy (1986); M. Bailey (pers. comm.)
N. lewisi	habitat alteration	NC	Stephan (1977); Bury et al. (1980)
Notophthalmus meridionalis	habitat destruction	TX	Judd (1985)

Table 2. Continued.

Taxon	Threats — Rarity	States	Information Sources
N. perstriatus	habitat destruction, drought	FL,GA	Christman and Means (1992); Dodd (1993); R. Franz (pers. comm.); Bury et al. (1980); Franz and Smith (1993); Dodd and LaClaire (1995)
Pseudobranchus striatus lustricolus	rare?	FL	Bury et al. (1980); Moler (1988, 1992c)
Pseudotriton montanus flavissimus	unknown	AL	Means (1986f)
P. ruber vioscai	unknown	AL	Means (1986g)
Siren intermedia	unknown	KY	Branson et al. (1981)
S. lacertina	rare?	AL	Mount (1986b)
Stereochilus marginatus	peripheral, forestry?	FL	Christman (1992); J. Palis (pers. comm.)
Typhlomolge rathbuni	habitat alteration	TX	Bury et al. (1980)
Typhlotriton spelaeus	pollution, collecting	AR	~~Reagan (1974)~~
Order Anura — Frogs			
Bufo houstonensis	habitat destruction	TX	Bury et al. (1980); USFWS (1984b)
B. quercicus	forestry	VA	Pague (1991a)
B. valliceps	rare?, peripheral	AR	Reagan (1974)
Hyla andersonii	habitat destruction	AL,FL,NC,SC	Means and Longden (1976); Mount (1980); Moler (1980, 1981); Cely and Sorrow (1983); Means (1986a, 1992a); Palmer (1977); Bury et al. (1980)
H. avivoca	unknown	AR, KY	Branson et al. (1981); Trauth (1992a)
H. cinerea	unknown	KY	Branson et al. (1981)
H. gratiosa	forestry, habitat destruction, rare?	KY,TN,VA	Echternacht (1980); Pague and Young (1991) Branson et al. (1981)
Pseudacris streckeri illinoensis	habitat destruction	AR	Bury et al. (1980); Trauth (1992b)
Rana areolata	unknown	KY	Branson et al. (1981)

Table 2. Continued.

Taxon	Threats — Rarity	States	Information Sources
R. capito aesopus	habitat destruction	FL,GA	Godley (1992); R. Franz, P. Moler, W. Seyle (all pers. comm.); Bury et al. (1980); Franz and Smith (1993)
R. c. capito	habitat destruction	GA,NC,SC	Braswell (1977b, 1993); S. Bennett, W. Seyle (both pers. comm.)
R. c. sevosa	habitat destruction	AL,FL,LA,MS	Bailey (1991); Dundee and Rossman (1989); M. Bailey (pers. comm.); Means (1986b)
R. okaloosae	rare?	FL	Moler (1985, 1992a)
R. palustris	peripheral	FL	Moler (1992b)
R. sylvatica	peripheral, drought	GA	C. Camp (pers. comm.)
~~*R. virgatipes*~~	~~peripheral, drought,~~ habitat destruction, pollution	~~FL,VA~~	Means and Christman (1992); Pague (1991b)
Scaphiopus bombifrons	rare?	AR	Trauth (1989)
S. holbrooki	rare?	WV	Pauley and Canterbury (1990)

[1] Affiliations of personnel contributing information: Mark Bailey (Alabama Natural Heritage Program); Steve Bennett (South Carolina Wildlife and Marine Resources Department); Carlos Camp (Piedmont College); Steve Christman (Quincy, FL); Richard Franz (Florida Museum of Natural History); Julian Harrison (Charleston Museum); Jeffrey McLemore (South Carolina Nongame and Heritage Trust Program); Paul Moler (Florida Game and Fresh Water Fish Commission); John Palis (Jonesboro, IL); Win Seyle (U.S. Army Corps of Engineers).

the status of amphibians throughout the United States.

In generally assessing historical information about amphibians residing in the Southeast, endangered and threatened amphibian accounts were written by individuals familiar with the biology of the species (see Table 2). However, few assessments were based on thorough studies and none included long-term quantitative data. Symposia have seemed to highlight more of what was not known about a species than what was known. Edited proceedings have usually contained information on life history, distribution, status, and threats, whereas journal articles contained little background data.

Concern about the status of particular herpetofauna species or communities also has resulted in many inventory programs, but much of this information remains unpublished and generally unavailable. For example, intensive herpetofaunal inventories based on quantitative sampling were prepared for Lake Conway, Florida (Bancroft et al., 1983), the proposed Cross Florida Barge Canal route in the Ocala National Forest, the proposed phosphate mining area in the Osceola National Forest, and the St. Marks National Wildlife Refuge in Florida. Unfortunately, reports of the results of such surveys are difficult to obtain and often lack crucial details concerning site locations, sampling methods and intensity, and statistical analysis. Herpetofauna inventories of various other national forests (e.g., Pearson et al., 1987), military reservations (e.g., Williamson and Moulis, 1979), and state and private lands also exist for areas scattered throughout the Southeast. Herpetofaunal inventories are presently under way at Eglin Air Force Base, Florida; Ft. Stewart, Georgia; and Camp Blanding, Florida.

Recent examples of single species amphibian surveys include *Rana capito* in North Carolina (Braswell, 1993); *R. capito* and *Notophthalmus perstriatus* in Florida (Franz and Smith, 1993); *N. perstriatus* in Georgia (Dodd and LaClaire, 1995); and *Ambystoma cingulatum* in Florida (Palis, 1992, 1993) (also see Table 2). All southeastern states now have Natural Heritage Programs to assemble data on declining species. Some of these programs are well advanced in data analysis (e.g., Florida), whereas others are just getting started (e.g., Georgia).

In the 1970s, ecological studies generally became much more intensive and quantified, and often integrated field and laboratory work to examine hypotheses of species interactions. Although they did not initially begin as monitoring studies, the ecological studies at the Savannah River Site (SRS) in South Carolina (see Gibbons and Semlitsch, 1991, and references therein) and Hairston's studies of terrestrial salamander competition (Hairston and Wiley, 1993) in the southern Appalachians are the only studies with truly long-term continuous data sets in the Southeast. Only the SRS study has data on all of the aquatic amphibian species in the local community.

Other studies are available covering a shorter time span. Dodd (1992) systematically monitored the amphibian community at a temporary pond in north Florida sandhills from 1985 to 1990. H. Mushinsky (University of South Florida) and A. F. Scott (Austin Peay State University) have quantitatively monitored the amphibian community on a Florida sandhill and in north-central Tennessee, respectively, since the early 1980s, although these data are not yet published. Delis (1993) compared amphibian community changes from the 1970s to the 1990s in an area of west-central Florida undergoing urbanization. Other studies have monitored a single species in a region or at a single locality for various amounts of time (M. Bailey, Alabama Natural Heritage Program; J. Palis, Jonesboro, Illinois; W. Seyle, U.S. Army Corps of Engineers, all pers. comm.), but the results of such monitoring are generally not available.

In conclusion, the literature on potentially imperiled amphibians in the Southeast is scattered and based on few quantitative studies. Much information remains unpublished or is otherwise unavailable. Therefore, it is often impossible to assess the accuracy or thoroughness of completed work. The only long-term data set on continuously monitored aquatic-dependent amphibians in the Southeast is available from SRS. At SRS, much annual variation occurs in the number of reproductive adults visiting a wetland. Reproductive output also varies annually, even when substantial numbers of adults reproduce (Pechmann et al., 1991).

There are numerous published studies on the ecology of individual aquatic-dependent southeastern species or groups of species. Locations used for such studies could serve as monitoring sites to assess the status of the species and habitat since the original studies were completed. Examples of some original assessments include the work of Trauth et al. (1992) assessing the status of an Arkansas population of hellbenders (*Cryptobranchus alleganiensis*) previously studied in the mid-1980s and Dodd's (1991) study of the Red Hills salamander (*Phaeognathus hubrichti*). Few follow-up assessment studies have been undertaken.

THREATS TO SOUTHEASTERN AQUATIC SPECIES

Amphibians that depend on aquatic environments in the Southeast potentially are vulnerable to a great variety of threats, although few detailed studies have specifically considered such problems within the region. The integrity of both aquatic and terrestrial habitats is important to amphibian survival, even among species that never venture beyond a single habitat type. Furthermore, the various life history stages (eggs, larvae, young, adults) may be differentially susceptible or sensitive to environmental perturbations. Studies that assess only one phase of a species' life cycle (e.g., surveys only of breeding habitat) may overlook important ecological requirements of other life history phases. Although we tend to discuss conservation in terms of individual species, an ecosystem approach that is sensitive to all life history phases is necessary to ensure the habitat integrity that ultimately will continue to support individual species.

Literature references to southeastern aquatic-dependent amphibians that currently might be in need of some degree of management are provided in Table 2. Habitat destruction and alteration are the most commonly identified factors affecting species' status. There are many cases where a species appears rare but is geographically peripheral to the region in question, or its true status is unknown. There is only one case where a "mysterious" decline may have occurred regarding an aquatic species. The salamander *Desmognathus auriculatus* appears to have declined or disappeared from sections of the Atlantic Coastal Plain in South Carolina and peninsular Florida (S. Christman, Quincy, Florida; J. Harrison, Charleston Museum, both pers. comm.), but no systematic surveys have been undertaken. However, populations of coastal plain desmognathine salamanders are known to fluctuate substantially in numbers from one year to the next (B. Means, Coastal Plains Institute, pers. comm.). Some specific threats to aquatic amphibians are discussed briefly below.

Habitat Destruction and Alteration

Even before the arrival of Europeans, Native Americans exerted considerable influence upon southeastern landscapes. Villages formed in circular patterns were interconnected by corridors and surrounded by considerable amounts of buffer land used for hunting

(Hammett, 1992). Lands were used for agriculture and large areas were burned to clear land and to improve hunting. After colonization by Europeans, land clearing and ecosystem modification accelerated and have culminated in the present frenzy to redesign the landscape.

The Southeast has been rapidly increasing in human population for several decades, and its metropolitan areas are among the fastest growing population centers in the United States. In Florida alone, where more than 9 million acres (3,642,300 ha) of wetlands already have disappeared (Cerulean, 1991), the population increases by a net 900 newcomers each day. In Arkansas, 6 million of the original 10 million acres (2,428,200 of 4,047,000 ha) of Mississippi Delta wetlands have been converted to agricultural land (Smith et al., 1984). In a west-central peninsular Florida study, species richness was less in urbanized areas than in nearby pristine areas, and temporary pond breeding species disappeared entirely from the urbanized site (Delis, 1993). Although vast areas have been cleared in the Southeast for agriculture, industry, and urban use, there is virtually no assessment of the landscape effects of land conversion on amphibian populations. It seems evident, however, that habitat changes (see papers in Hackney et al., 1992; Boyce and Martin, 1993), and with them changes in aquatic amphibian populations, have been enormous.

Habitat alteration may occur without obvious large-scale topographic changes. For example, a massive boom in human population on the Edwards Plateau of Texas has increased the withdrawal of ground water from the Edwards Aquifer. As more and more water is withdrawn, water tables have decreased. In the future, springs and streams in this region are likely to dry completely, especially during periods of drought. This situation could lead to the loss of a unique aquatic biota that includes spring and cavernicolous salamanders (U.S. Fish and Wildlife Service, 1984a; Chippindale et al., 1993).

Habitat Fragmentation

Although habitat fragmentation affects biota in different ways (e.g., Mader, 1984), land use patterns resulting in fragmentation can influence amphibian population genetic structure (Reh and Seitz, 1990). Amphibian populations are most abundant when there is a mosaic of habitats located within a regional landscape (Mann et al., 1991). In such a context, metapopulations may develop which result in a dynamic equilibrium through time. However, if populations become overly fragmented, emigration and immigration may be inhibited or stopped, thus preventing recolonization from source populations. The effect of fragmentation on amphibians depends on the degree of isolation (Sjogren, 1991a). Small, isolated populations are particularly susceptible to environmental perturbations (Sjogren, 1991b) and to stochastic variation in demography that can lead to extinction even without external perturbations (Lande, 1988; Pimm et al., 1988). Isolation by habitat fragmentation thus becomes a threat to the regional persistence of species.

Forestry Practices

Most discussions of the effects of forestry on amphibians in the Southeast focus on salamanders in clearcuts (Blymer and McGinnes, 1977; Ash, 1988; Dodd, 1991; Petranka et al., 1993; Ash and Bruce, 1994; Petranka, 1994), although a few recent studies have examined amphibian communities in the coastal plain (Phelps, 1993; Dodd, 1995b; O'Neill, 1995; Phelps and Lancia, 1995; Means et al., 1996). Clearcutting reduces salamander populations because it eliminates shade, reduces forest litter (especially if litter is piled and burned), increases soil temperature, reduces soil moisture, and destroys wetlands. Herbicides are frequently used in such operations, yet little is known of their effects on amphibians (but see Bidwell and Gorrie, 1995).

Depending on the type of site preparation, clearcutting practices also reduce or eliminate burrows and other hiding places needed by aquatic habitat-dependent amphibians when they are away from their breeding sites. For example, clearcutting an area adjacent to an *Ambystoma talpoideum* breeding pond in Louisiana lowered the survivorship of immigrating adults using the clearcut site and displaced other adults to less suitable habitat (Raymond and Hardy, 1991). Other attributes which affect amphibian persistence after timber cutting include the status of amphibian populations prior to cutting, the type of cut (selective vs clear), the type of forest replanted, the size of the cut, the amount of time since last cut (Grant et al., 1994), and the distance to the nearest source populations. Mature southeastern pine plantations also support far fewer amphibians than adjacent deciduous forests (Bennett et al., 1980). In Florida sand pine scrub, clearcutting seems to mimic intensive wildfire; the richness and diversity of amphibians appear more dependent on the nearest water source for breeding than on the type of disturbance (Greenberg, 1993). However, clearcutting reduced amphibian species abundance in pine flatwoods tenfold by adversely affecting reproductive success (Enge and Marion, 1986). Regarding the effects of timbering, stream-dwelling species and their larvae have received little attention in the Southeast, although adverse effects to stream-dwelling amphibians caused by logging in the Pacific Northwest are well documented (Bury and Corn, 1988; Corn and Bury, 1989).

On the southeastern Coastal Plain, vast pine plantations have replaced the native longleaf pine (*Pinus palustris*) savanna. During planting and site preparation, much of the land was ditched in an effort to speed water runoff. Literally thousands of acres of wetlands disappeared or were substantially altered. Ditching occurred between ponds to facilitate water transfer; water essentially flowed downhill, although slowly, thus reducing available hydroperiods for amphibian larval development. A second type of ditching occurs around wetlands. Circumferential ditching results in lowered water tables with concomitant vegetative changes, thus drastically altering or eliminating hydroperiods. Unditched ponds are more persistent than ditched ponds, and have greater amphibian species richness during dry periods (Harris and Vickers, 1984; Vickers et al., 1985). In addition, more aquatic amphibian species are associated with unditched ponds. This is especially important because many temporary pond-breeding amphibians exhibit breeding site fidelity and other obligate breeding requirements which can be impacted by ditching.

The loss of the longleaf pine forest on the coastal plain of the southeastern United States has been dramatic (Means and Grow, 1985; Noss, 1989; Boyce and Martin, 1993; Stout and Marion, 1993; Ware et al., 1993). Concern for the survival of the coastal plain forest in Georgia was expressed at least as early as 1906 because of logging, turpentining, and land clearing for agriculture and "civilization" (Harper, 1906). Since the 1940s, old-growth longleaf pine forest has been converted to slash (*P. elliottii*) and loblolly (*P. taeda*) pine plantations throughout the Southeast. In southeast Georgia, for example, longleaf pine declined 36 percent between 1981 and 1988 to 230,000 acres (93,081 ha; see Johnson, 1988), whereas in southwest Georgia, longleaf pine declined four percent during these same years to 205,000 acres (82,963 ha)(Thompson, 1988). Today, less than one percent of the old growth longleaf pine forest remains (of the more than 70 million acres

[28,329,000 ha] present when Europeans colonized the continent). Most remaining forest is scattered and poorly managed. Even in national forests, longleaf pine has declined substantially (Means and Grow, 1985). In the last few decades, drastic changes probably have occurred in the composition and structure of the amphibian community in regions that formerly held longleaf pine (Dodd, 1995b; Means et al., 1996), but no baseline data exist to document the effects of this continuing massive landscape alteration.

Mining

Extensive coal strip mining is carried out in West Virginia, Virginia, Kentucky, Tennessee, and Alabama. In many instances, mining occurs directly through small streams or ponds, and mine tailings are pushed into the larger rivers. In Florida, vast areas have been strip-mined for phosphate. Mining not only destroys aquatic amphibian habitats outright, it also results in toxic pollution, decreased pH, and siltation of streams and rivers. Low pH combined with high levels of conductivity (an indication of the presence of pollutants) limit the presence of larval salamanders of the genus *Desmognathus* in mine-affected streams of the Cumberland Plateau (Gore, 1983). Paradoxically, amphibians have bred in strip mine ponds as long as the pH was not too low and toxic waste was prevented from entering the pond (Turner and Fowler, 1981; Lacki et al., 1992).

Transportation Corridors

Transportation corridors, especially roads, can have serious deleterious effects on amphibian populations (Langton, 1989). Road construction can lead to habitat destruction in both terrestrial and aquatic environments, and can negatively alter breeding habitats through increased siltation. Increased siltation can lead to increased amphibian mortality because of its own secondary effects. For example, nearly all aquatic life was eliminated downstream after U.S. Highway 441 was rebuilt in 1963 in the Great Smoky Mountains National Park. Toxic substances associated with leachates from roadfill were suspected as the cause. Laboratory experiments confirmed that roadfill leachates were toxic to larval shovel-nosed salamanders (*Leurognathus marmoratus*). The major components of the leachate responsible for toxicity included low pH combined with high heavy metal concentrations (Mathews and Morgan, 1982).

Roads may separate overwintering sites from breeding sites and increase mortality as animals attempt to cross. For example, Heine (1987) demonstrated that 26 vehicles per hour on one road was enough traffic to ensure that no toads successfully crossed. Road construction also can lead to habitat fragmentation, and in doing so can hinder immigration and emigration, and isolate populations (Laan and Verboom, 1990) leading to deleterious effects associated with small population size (Sjogren, 1991b). Furthermore, the noise levels and artificial lights associated with traffic may disrupt breeding activities. Noise makes it difficult to hear conspecifics or causes frogs to completely stop calling (author's pers. obs.). Bright artificial lighting can adversely affect frogs' abilities to detect and consume prey (Buchanan, 1993).

Climate Change

If climate changes, possibly in response to increasing levels of greenhouse gases, then there are bound to be changes in the diversity of southeastern amphibians. Most of our endemic species and species-rich amphibian communities are found on the higher elevations of mountains in the cool southern Appalachians and Ozarks, or in specialized coastal plain habitats, such as temporary ponds. Spring adapted salamanders of the Edwards Plateau are sensitive to alterations in ground water levels. These species would be particularly susceptible to climate changes which alter rainfall patterns or elevate mean annual temperatures. However, the potential for changes in amphibian diversity seems to have been overlooked in the climate change debate. For example, amphibians are mentioned, briefly, only twice in 26 chapters of a recent book which examines the effects of global warming on biological diversity (Peters and Lovejoy, 1992).

pH

The acidity of aquatic habitats can play a major role in limiting the distribution of amphibians. Decreased levels of pH in aquatic habitats may result from acidic precipitation or point-sources of pollution, such as abandoned mines. Acid concentration may increase steadily or come in pulses, such as during heavy rains or from snow melt. Although the Southeast has not experienced as many problems from acid rain as other parts of the United States, the acid content of our precipitation is increasing (Haines, 1979). For example, H+ increased 19-fold from 1955 to 1979 in Great Smoky Mountains National Park (Mathews and Larson, 1980). Bioassay results suggested that pH levels were near toxic to larval shovel-nosed salamanders, although not as toxic to adults (Mathews and Larson, 1980).

The literature on the effects of pH on amphibians is voluminous and complex (Freda, 1986; Dunson and Wyman, 1992). Low pH has different effects on different species of amphibians and, indeed, there may be intraspecific differences in pH sensitivity that varies geographically. Furthermore, these intraspecific differences may or may not have a genetic basis (Pierce and Wooten, 1992). In general, the eggs and developing larvae are the most sensitive life stages to low pH (< 4.5). A low pH alters the cellular chemical environment by disrupting the Na+ and Cl- balance both in terrestrial (Frisbie and Wyman, 1991) and aquatic life stages (Freda and Dunson, 1984). This, in turn, affects salamander spatial distribution since salamanders avoid soils of low pH (Wyman and Hawksley-Lescault, 1987; Freda and Taylor, 1992). Low pH also may impair the vitally important chemosensory system of amphibians (Griffiths, 1993) and inhibit larval feeding (Roudebush, 1988).

Low pH can also have indirect effects which can kill eggs, larvae, or even adults (Sadinski and Dunson, 1992). A low pH acts to inhibit amphibian egg capsule enlargement, and thus limits the space available to the growing embryo. In addition, high acidity inhibits proper jelly formation. Jelly allows spacing of the eggs within an egg mass which ensures that each developing embryo has an adequate oxygen supply (Seymour, 1994). If jelly does not form properly, death from anoxia results. Chronic or intermittent low pH also can disrupt environmental trophic interactions (Sadinski and Dunson, 1992), and can lead to problems associated with long-term environmental stress. For example, phytoplankton which are fed upon by tadpoles are also sensitive to low pH (Haines, 1981).

Toxic Substances

A great many substances are likely toxic to amphibians, at least during part of their life cycle. Toxicants need not be synthetic chemicals. For example, salt spread on roads during winter can affect the chemistry of amphibian breeding sites. Toxic chemicals can enter the environment in many ways, both intentionally and accidentally. There have been numerous instances of inadvertent release of toxic materials into aquatic habitats because of highway or railroad accidents.

Surprisingly little research has been done on the effects of toxic chemicals on amphibians, and even then most work has focused on only one part of the life cycle. Examples of toxic materials known to adversely affect amphibians include heavy metals (aluminum, mercury, selenium), pesticides (toxaphene, heptachlor, malathion, endrin, methoxychlor), herbicides (DEF, trifluralin, atrazine), fungicides (furanace, malachite green), phenols, carbon tetrachloride, and nitrite. Literature summaries are provided in Birge et al. (1980), Power et al. (1989), and Hall and Henry (1992).

Data on the level of toxic chemicals in wild populations of amphibians, much less those of the Southeast, are nearly non-existent. However, Hall et al. (1985) noted metabolites of DDT as well as PCBs (primarily chlordane constituents) in *Necturus lewisi* from the Tar and Neuse rivers, North Carolina. The herbicide atrazine was implicated as contributing to large frog (*Rana pipiens*) die-offs in Wisconsin (Hine et al., 1981).

In addition to direct effects, certain toxicants may affect amphibians differently depending on pH. For example, aluminum has adverse effects upon amphibians, but the level of adversity differs depending on species, life stage, and pH (Beattie and Tyler-Jones, 1992; Bradford et al., 1992; Jung and Jagoe, 1993). Lowered pHs amplify the toxicity of heavy metals to amphibians.

Endocrine Mimics

Many chlorinated chemicals (DDT, PCBs, etc.) have been dumped in huge quantities into the environment during the 20th century, and as they travel throughout food chains they become magnified in concentration. Chlorinated chemicals can act to impair development, block intracellular communication, and induce enzymes that break down hormones. In addition, many of these persistent compounds function, even in minute quantities, as hormones, especially mimicking estrogen. Some of the side effects of endocrine mimics are thyroid dysfunction, metabolic abnormalities, decreased fertility, birth deformities, abnormal sexual development, and immunosuppression (Carey and Bryant, 1995; Stebbins and Cohen, 1995). Although no specific examples exist yet for amphibians, xenobiotics have been implicated in partial sex reversals and gonadal feminization in a wild Florida population of American alligators (*Alligator mississippiensis*) (Guillette et al., 1994).

Amphibians are likely to be especially sensitive to the action of endocrine mimics because they are in close direct contact with chemicals in their environment, and the amphibian skin and egg capsule are highly permeable. Because hormones normally function in minute quantities and are vital to normal development (Hourdry, 1993), susceptibility to xenobiotics could be devastating during the complex changes that occur during hormonally-induced amphibian metamorphosis.

Ultraviolet-B Radiation

Recent evidence suggests ultraviolet-B (UV-B) radiation has adverse effects on amphibian larval hatching success and that sensitivity to UV-B varies among species (Blaustein et al., 1994a) or is exacerbated by low pH (Long et al., 1995). Species with high levels of photolyase (e.g., *Pseudacris* spp.), an enzyme involved in DNA repair of ultraviolet radiation damage, are less prone to the adverse effects of UV-B radiation than species with low levels of photolyase (e.g., *Bufo* spp., *Rana* spp.). Many populations of *Bufo* spp. and *Rana* spp. have declined in the western United States, whereas *Pseudacris triseriata* populations have not. Frog embryos (*Rana clamitans* and *R. sylvatica*) exposed to high levels of UV-B had higher rates of developmental abnormalities and increased mortality than controls which were shielded from UV-B (Grant and Licht, 1993). UV-B also can have detrimental effects on embryo growth. UV-B radiation has increased recently in the northern hemisphere because of ozone depletion (Blumthaler and Ambach, 1990; Kerr and McElroy, 1993). If UV-B adversely affects southern Appalachian anurans, toads (*Bufo* spp.) and true frogs (*Rana* spp.) would seem most likely to be affected.

Exotics, Predators, and Competitors

There is no literature on the effects of the many exotic fishes in southeastern waters on native herpetofauna. Fish may be both competitors and predators of amphibians, depending on life cycle stage (Bristow, 1991). They have been implicated in declines of western amphibians both as predators (Bradford, 1989) and as disease vectors (Blaustein et al., 1994b). Stocking of predatory fishes in ponds previously free of fish undoubtedly leads to a change in the amphibian community because many amphibians are defenseless against fish predators. Conversion of temporary ponds to permanent ponds by digging and blasting, followed by fish introductions, often leads to a loss of the temporary pond breeding species. The effects of exotic frogs, especially the marine toad (*Bufo marinus*) and Cuban treefrog (*Osteopilus septentrionalis*), on native amphibians are unknown, although anuran species richness was reduced in at least one area having marine toads, compared to a similar area without them (Rossi, 1981). Release of other exotics undoubtedly occurs with unknown effects. One south Florida tropical fish dealer reported selling 50,000 eastern newts in Florida that originated from outside the state (Enge, 1991). Many of these exotic newts undoubtedly were released intentionally or unintentionally.

Birds and mammals also may exact a substantial toll on amphibian populations, especially exotic cattle egrets, armadillos, and wild hogs. In addition, populations of some native species, such as raccoons, may become so large because of a lack of natural predators and adaptation to human surroundings that they in turn reduce amphibian populations beyond normal levels. The overabundance of some native species is an issue which biologists are only beginning to confront (Garrott et al., 1993).

Finally, there are few data on the effects of exotic invertebrates, especially imported red fire ants (*Solenopsis invicta*), on native amphibians. Ground-dwelling vertebrates are especially sensitive to this ravenous predator (Mount, 1981), and fire ants have been reported to kill endangered Houston toads (*Bufo houstonensis*) as they metamorphose (Freed and Neitman, 1988). Fire ants are especially abundant in the moist perimeter surrounding ponds and lakes, and they can float in mats across ponds from vegetation clump to vegetation clump. Fire ants have few predators and have expanded their range throughout the Southeast.

Collecting

Collecting specimens for the pet trade or biological laboratories probably has had some impact on local amphibian populations, but few data are available. Trauth et al. (1992) suspected that collection of hellbenders in the Spring River, Arkansas, contributed to observed population declines. From 1 July 1990 to 30 June 1991, 804 salamanders and 18,170 frogs were collected legally for the Florida pet trade (Enge, 1991). Included were 5,066 *Hyla cinerea*, 3,265 *Bufo terrestris*, 2,674 *Hyla gratiosa*, and 249 *Siren lacertina*. In 1992, 246 salamanders and 23,019 frogs were collected and sold in the pet trade in Florida

(Enge, 1993). Most sales went to New York, Pennsylvania, and Tennessee. Concern for the effect of biological supply house collection on frog populations is not new (Gibbs et al., 1971). In the early 1970s, U.S. frog suppliers shipped 9 million frogs (over 326,000 kg) per year. The number of frogs shipped by southeastern supply houses is unknown.

Loss or Decline of Associates

If species that are preyed upon by amphibians decline or disappear, amphibian populations may be expected to follow suit. The use of pesticides and the influence of toxics, pH, and habitat alteration all may be expected to affect amphibian prey populations. In addition, amphibians sometimes rely upon the burrows of other species for shelter when they are away from ponds. If these associated animals are eliminated, fewer shelters may be available. A few amphibians inhabit the burrows of specific associates. For example, gopher frogs (*Rana capito*) nearly always reside in sympatric gopher tortoise (*Gopherus polyphemus*) burrows when the frogs are not at breeding ponds. Yet, the number of gopher tortoises is estimated to have declined by 80 percent during the last 100 years (Auffenberg and Franz, 1982). The effect of the decline of tortoises and their sheltering burrows on gopher frogs is unknown.

Drought, Cold, and Disease

Drought, cold, and disease are natural factors that affect amphibian communities (e.g., Dodd, 1993, 1995a). Drought can lead to localized extirpation. Excessive cold can induce winterkill in torpid amphibians. Disease can wipe out populations. However, the chronic effects of these factors on amphibian populations, if any, remain unknown. Under pristine conditions, amphibian populations often may expand and contract in response to such natural variables affecting local distribution, thus forming a dynamic equilibrium (Sjogren, 1993a, 1993b). Under present human-dominated landscapes, however, populations may be so fragmented or under such a variety of stresses that they are unable to rebound from extrinsic environmental factors causing periodic population fluctuations. If many amphibian populations function as metapopulations, the long-term survival of local populations might be jeopardized by isolation from source populations coupled with natural environmental fluctuations.

In fact, "natural" factors may not be as natural as they first appear. For example, droughts may result from global climate change or they may be magnified by habitat alteration such as deforestation or overgrazing. The effects of disease also might be facilitated by human activity. Carey (1993a, 1993b) has proposed a model whereby sublethal stress (such as that associated with chronic low but sublethal pH, or high concentration of a toxicant, or increased UV-B radiation) induces either direct or indirect immunosuppression because of the prolonged elevation of adrenal cortical hormones. Depressed immunity makes the animal more prone to naturally occurring pathogens, such as red-leg disease causing bacteria (*Aeromonas* spp.), especially during periods of torpor. This model is consistent with observations on declining amphibians in many Rocky Mountain populations where amphibian populations have been known to decline gradually and then one year simply fail to emerge from hibernation.

A pathogenic fungus has been implicated recently in the decline and disappearance of *Bufo boreas* in the western United States (Blaustein et al., 1994b). The fungus (*Saprolegnia ferax*) is circumglobal in distribution and commonly occurs on fish. However, fish are not native to the high mountain habitats occupied by *B. boreas*, and the pathogen is thought to have been introduced when trout and salmon were stocked in high mountain streams and lakes. The same fungus has extirpated other frog populations in the U.S. and Europe (for a review of this topic see Blaustein et al., 1994b). Although the extent of amphibian fungal infections is unknown in the Southeast, every egg mass (*Rana* sp.) I examined during March 1994 in several ponds on Trail Ridge in southern Georgia was infected by an as yet unidentified fungus.

SUMMARY

The southeastern United States holds a rich temperate amphibian assemblage containing a great degree of endemism. Endemic species are especially well represented among the salamanders. A varied topography and complex geologic history have provided the necessary conditions that have resulted in this region's high level of speciation. However, the amphibians of this area, and particularly the fully aquatic species, face a multitude of threats to their long-term existence. These threats generally do not act independently, but instead act in concert to have potentially serious long-term effects. Many amphibian species have been identified as needing conservation programs and management, but few scientific studies have assessed direct effects to species or ecosystems. There also are few studies detailed enough to show trends or to separate unnatural trends from normal population fluctuations.

Although natural population fluctuations undoubtedly exist, it is extremely naive and certainly not objective to call simply for "more monitoring" (Pechmann and Wilbur, 1994). At a time when conservation related funding is nearly nonexistent and no agencies seem able to initiate long-term monitoring on a scale required to assess wide-ranging threats to amphibians, the call for more monitoring seems an effective mask for doing nothing. How can interest be generated in monitoring "common" or non-threatened species, much less communities and ecosystems, when programs directed at the conservation of critically endangered species are under-funded or not funded at all? Given the cumulative assaults on the biosphere in the late 20th century, I suggest Chicken Little is better in tune with biological and political reality than Nero with his fiddle (see Blaustein, 1994). Rome, after all, burned.

As Gibbons (1988) has discussed, a new attitude is needed toward the recognition of the importance of amphibians to ecosystem functioning. No longer can these species be assigned a role of non-importance in wildlife and land management. Attention must be focused on threats to species inasmuch as these threats may be symptomatic of serious environmental problems. We need to study the seemingly common species (Dodd and Franz, 1993), as well as the rare or endangered species. Our casual perceptions may not always give an accurate assessment of population status. Finally, we need an ecosystem, landscape, and watershed approach to understanding the role of amphibians in imperiled aquatic systems as well as adequate funding from private and governmental agencies (Mittermeier et al., 1992) to carry out necessary research and management programs.

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